Evaluating baseline conditions and resulting changes in demersal fish communities of South East Australia

Novaglio Camilla

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Institute for Marine and Antarctic Studies

University of Tasmania

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Statements and declarations

Declaration of Originality

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Camilla Novaglio

Hobart, November 2015

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Dedicato alla mia cara nonna Lina.

Abstract

Knowledge of the full extent and severity of ecological changes in human-influenced ecological systems is needed to identify today's management priorities and to set realistic restoration objectives for the future. However, reconstructing ecosystem baselines and understanding the causes and rates of ecological changes is often hampered by the scarcity of data about population and community status before exploitation, and by the challenges of fitting these (historical) data into modern analytical methods.

This study aimed to identify the main sources of fishing impacts on marine communities of South East Australia and to gather and examine historical data available for the region that can provide information on baseline (pre-fishing) conditions to compare changes in fish communities as the fishing industry developed. South East Australia provides an ideal case study as its history of exploitation is relatively recent and there were surveys undertaken prior to exploitation.

While a range of commercial fisheries have evolved in South East Australia since European colonization, during the last century bottom trawling has been the major fishing activity in the region. Bottom trawl surveys that included information on demersal species abundance and community structure were carried out both before and at different stages since trawling exploitation begun, and thus provide insights into the full extent of fishing impacts on South East Australian demersal fish communities.

These surveys, covering the period 1898-1997, were performed by various research agencies, which collected and organized catch and effort data in different formats.

Additionally, the detail of the information reported changes across surveys and over the years. Hence, the initial need was to collect, digitalize and standardize all the information available. The bottom trawl survey dataset resulting from this step contained a total of 3,083 tows sampling 574 species among chondrichthyes and osteichthyes. It spans the entire history of trawling exploitation and is analyzed in this study as an entire dataset for the first time.

A comparison of pre- with post-trawling exploitation data (1898-1910 and 1980s-2010s, respectively) revealed marked changes in the structure of demersal fish communities of South East Australia. These included shifts in the catch composition of the main families, as well as sharp declines in the total and individual family catch rate, most likely related to the effect of fishing. Among the steepest declines were those of key commercial families, such as flatheads and morwongs, on the continental shelf of Tasmania.

The effect of trawling on demersal fish communities of South East Australia was also revealed by the application of species accumulation curves to the survey dataset. Specifically, the rate of species accumulation with area decreased as trawling intensity increased, suggesting that trawling modified community structure through the removal of particular species and through changes in the abundance and spatial distribution of the remaining species.

This study’s findings have direct application to management and monitoring of the natural resources in South East Australia, and important implications for sustainable use and conservation prioritization. The study also provides a framework and approach that can be of guidance for the collection, standardization and analysis of analogous, patchy, unbalanced and overlooked historical datasets around the world.

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1 Chapter 1 – Introduction

Fisheries have contributed to food production, employment and revenue in many countries and communities over the centuries. Small populations of early humans depended on the extraction of marine resources for survival (Attenbrow, 2010) and, since then, the importance of fisheries for societies, their sustenance and economy has grown dramatically (Roberts, 2007). Today the fishing industry contributes to about 17% of human consumption of animal protein and provides tens of thousands of jobs globally, thus playing a key role in food security and human wealth (FAO, 2014). This role is likely to remain essential as the world population is expected to increase from the present 6.8 billion to about 9 billion by 2050 (UN-DESA, 2009), and both food production and employment are of greater concern than ever before (FAO, 2014).

Despite these benefits, fishing also impacts marine ecosystems and has the potential to alter their capacity to provide benefits now and into the future (Jackson & Johnson, 2001; Myers & Worm, 2003; Pandolfi *et al.,* 2003; Lotze *et al.,* 2006; Halpern *et al.,* 2008). Fishing has both direct and indirect effects on population structure and ecosystem function. Populations and ecosystems are impacted mainly through the selective removal of target species, through the by-catch of non-commercial species, and through habitat modification (Dayton *et al.,* 1995; Jennings & Kaiser, 1998; Kaiser, 1998; Watling & Norse, 1998). For instance, overexploitation of target species can reduce their abundance, spawning potential and alter population parameters, as well as impacting their associated and dependent species. Further, the substantial removal of top predators, often the most sensitive to fishing, can modify trophic structure and the flow of biomass across the ecosystem (Pauly *et al.,* 1998; Stevens *et* *al.,* 2000; Ferretti *et al.,* 2013). Hence high levels of fishing exploitation may also undermine ecosystem integrity and productivity.

The challenge in managing fisheries is to safeguard social and economic benefits while ensuring acceptable levels of impacts on marine ecosystems (noting that even low levels of fishing will have some impact), so that natural resources are preserved and maintained for future generations, and the long-term viability of the fishing industry is secured (Jennings *et al.* 2014). A first requirement for sustainable exploitation of marine resources is an understanding of the current status of marine populations and ecosystems and the level of exploitation they can sustain (Hilborn & Walters 1991). This chiefly depends on the extent and magnitude of past impacts (Jackson *et al.,* 2001, 2011). For instance, resources that had been extensively harvested in the past may be currently depleted and no longer able to provide the optimal benefits that they could.

Historical ecology is a discipline that studies past interactions of human societies with natural systems to understand current biotic conditions. It is an interdisciplinary approach that links the humanities and the natural sciences, and involves a wide range of research fields (e.g. history, ecology, anthropology, paleontology and archeology) all adding to the understanding of long-term changes in human-influenced natural systems (Balée, 2006; Ferretti *et al.* 2014; Szabó, 2014).

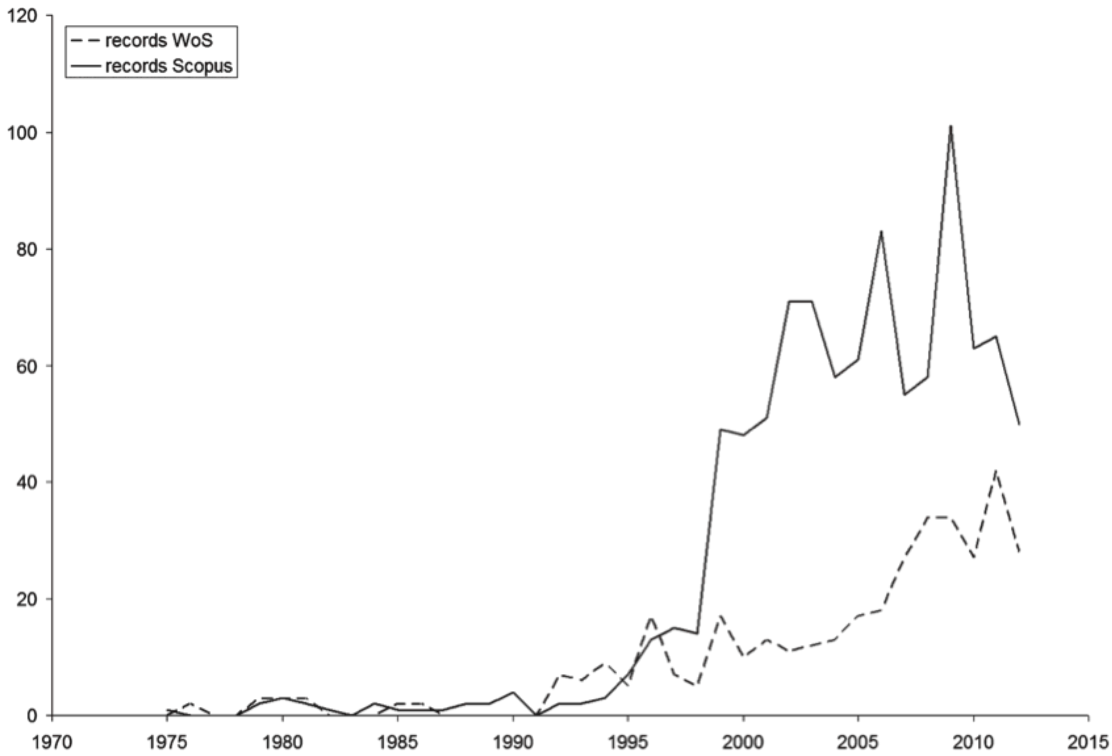

Figure 1-1. Number of paper that defined themselves as 'historical ecology' in the *Web of Science* (WoS) and *Scopus* databases from 1975 to 2012 (Szabó, 2014).

Historical ecology was born in Europe in the 1960s, but since the 1990s interest in this discipline has soared globally (Fig. 1-1, Szabó, 2014) and expanded into the marine realm (Pauly, 1995; Jackson *et al.* 2001, 2011; Pitcher, 2001; Pinnegar & Engelhard, 2008). Accordingly, the concept of 'shifting baselines' of fisheries was introduced in 1995 (Pauly, 1995), and refers to the way changes in marine populations or ecosystems are misinterpreted due to the significant difference of the reference baseline from the 'pristine' state of the system, and calls for assessment of ecosystems over time periods that align with human impacts on marine resources.

The discipline of marine historical ecology is closely linked to that of Marine Environmental History (MEH) (see Holm *et al*. 2001 for a reference to MEH). Whereas the first has a primary ecological focus, the second tackles the problem of “what has happened” using a more historical approach. They share interests, such as the discovery and interpretation of long-term fisheries data, and are often complementary, so that one could hardly exist without the other. Both call for deeper knowledge about past fishing practices and related changes to marine populations and ecosystems to better understand today’s Oceans and societies, thus both inspired this study.

Since the 1990s, there have been important efforts made to investigate the original (pre-fishing) state of exploited marine ecosystems using many forms of historical data, including anecdotes and oral histories (Sáenz–Arroyo *et al.* 2005; Saenz-Arroyo *et al.* 2006; McClenachan & Cooper, 2008; Montes *et al.*, 2008), fish guides and photographs (McClenachan, 2009; Last *et al.*, 2011), newspaper articles (Thurstan *et al.*, 2014), archaeological evidence, old literature and fisheries statistics (Lotze & Milewski, 2004; Lotze *et al.*, 2006; Thurstan *et al.,* 2010), fishery logbooks (Ferretti *et al.,* 2008; Alexander *et al.,* 2009), and data from long-term scientific surveys (Baum & Myers, 2004; Ferretti *et al.,* 2013). More recently, global initiatives, such as the History of Marine Animal Populations (HMAP, 2001) and the Sea Around Us programs, have promoted and coordinated research efforts worldwide. Together these studies strongly suggest that significant structural, and thus functional, changes have occurred worldwide over many centuries, to the point that today hypotheses about the structure and dynamic of marine ecosystems can’t be regarded as realistic without a good knowledge of the past situation. They also support the importance of historical data, not only to clarify underlying causes and rates of ecological changes, but also to develop new strategies for mitigation and restoration of ecosystems that are unlikely to emerge based on the limited perspective provided by recent observations alone.

Although essential, reconstructing ecosystem baselines and understanding the causes and rates of ecological change are difficult tasks. They are often hampered by the scarcity of historical data, the effort required to compile and convert such data to a format suitable for analysis, and the need to develop methods of analysis that overcome issues related to the comparability of historical with more recent data. For instance, sampling focus and design, as well as data quality and resolution (e.g. taxonomic resolution) may vary in time as a result of changes in knowledge, research priorities, and skill and technological improvements, thus complicating comparisons. Due to the challenges of fitting historical data into modern analytical methods, such as assessments of changes to fish stocks and communities, this information has often been omitted (McClenachan *et al.,* 2012; Ferretti *et al.,* 2014).

To overcome some of the limitations intrinsic to the analyses of historical data and to ensure the reliability of the results, a 'best practice' approach has been described (Swetnam *et al.,* 1999; McClenachan *et al.,* 2015). This approach follows five steps: 1) define the spatial and temporal scale of the process thought to be responsible for changes in populations or ecosystems (e.g. the spatial and temporal scale of a fishery); 2) identify a full range of information sources relevant to the research question; 3) address biases in the historical records; 4) design analyses to deal with data gaps; and 5) use multiple lines of evidence.

Following this approach, the present thesis attempts to reconstruct ecological baselines for demersal fish communities of South East Australia and to assess changes in these communities since the beginning of commercial fishing. I focus on fish communities of South East Australia because in this region the history of commercial fishing is relatively short and data on fish abundances and community composition were collected before the development of the fishing industry, and at various times subsequently. This chronology provides a rare window of observation that may have been lost in other regions of the world where exploitation began long ago and was followed (instead of preceded) by scientific investigation, so that ecosystem changes cannot be fully tracked. In this study, I mainly focus on the effects of fishing rather than on those of other potential drivers of community change (Halpern *et al.,* 2008), such as coastal development (Lotze *et al.,* 2006), water pollution (Kemp *et al.,* 2005), and climate change (Perry *et al.,* 2005), because I maintain that fishing may have had the major impact on the fish communities considered here. These are offshore communities of the continental shelves and slopes, thus relatively little impacted by coastal development and water pollution, but consistently exploited by commercial and, more recently, recreational fisheries (Tilzey & Rowling, 2001; Henry & Lyle, 2003; Woodhams *et al.,* 2011). In addition, the South East Australia region is particularly dry, with little runoff from land (Young *et al.,* 1996), so that water pollution from agriculture and mining industries may be limited. On the other hand, aware that climate change may have had a noticeable impact on these communities (Hobday, 2011; Johnson *et al.,* 2011; Last *et al.,* 2011), I consider some of its possible effects, but to a lesser extent.

1.1 Thesis structure

An understanding of the history of resource exploitation provides insights into the spatial and temporal scale of human impacts on natural communities, and points to the main sources of these impacts. Chapter 2 reviews the history of fishing exploitation and management in South East Australia, with a major focus on bottom trawling. This chapter also identifies historical data that could be informative about long-term changes in fish species abundances and fish community composition in this region, and provides the basis for selecting data from scientific bottom trawl surveys carried out along the South East Australian coast between 1898 and 1997 for analysis in chapters 4 and 5.

The resolution and quality of the data identified in chapter 2 vary from survey to surveys and in chapter 3 the focus is on data compilation, standardization and identification of gaps and limits. This included sourcing the data, seeking out and transcribing old records and reports, looking for missing information (e.g. nets' characteristics and species names used at the time of data collection), and convert data gathered from different sources to a common format, suitable for analysis.

Chapters 4 and 5 provide two contrasting methods of interrogation and analysis of the trawl survey datasets that can overcome some of the complications highlighted in chapter 3. Specifically, chapter 4 compares sets of data collected before (i.e. 1898 and 1910) and after (i.e. 1980s and late 2000s) the onset of the trawl fisheries in the region and identifies broad patterns of changes in demersal fish communities and fish families. In a different approach, chapter 5 tests the application of species accumulation curves to the trawl surveys dataset to examine the impact of trawling on demersal fish community structure. Although species accumulation curves have been widely used in terrestrial ecology, this chapter tests for the first time their use as an index of human impacts on marine communities, using data available from long-term bottom trawl surveys. The choice of species accumulation curves is justified by their limited data requirements (i.e. list of species caught and area sampled), which in South Eastern Australia is available for all surveys carried out between 1976 and 1997. Therefore, the chapter considers this specific set of data.

Chapter 6 provides a general discussion of the key findings and implications of this study, and proposes directions for future research. Reconstructing the baseline (pre-fishing) structure of South East Australia demersal communities and quantifying the impact of trawling intensity on these communities helped to understand causes and rates of ecological changes, with direct applications to current assessment and management of natural resources in this region. Further, this thesis provides a framework that can be of guidance for the collection, standardization and analysis of analogous, patchy, unbalanced and overlooked historical datasets around the world.

Chapters 2, 4 and 5 are written in a style suitable for publication. Whilst I have attempted to maintain a logical flow of ideas throughout the thesis, these chapters can be read independently.

2 Chapter 2 - Historical overview of fishing exploitation and scientific bottom trawl surveys in South East Australia

2.1 Abstract

The importance of an historical perspective in fishery science has been widely recognized. Knowledge of past fishing practices and exploitation rates can reveal the full extent and severity of ecological changes, thus informing today's management priorities and identifying realistic restoration objectives. However, historical information is often buried in archives and old reports. Here I bring together archived reports and scientific documents to provide a review of the history of fishing exploitation in South East Australia from the extraction of marine products by Australian Aborigines to the development of the modern trawl fishery, and I identify historical data that can help inform long-term changes in fish species abundances and fish community composition in this region.

2.2 Introduction

Fisheries have contributed to food production, employment and revenue in many countries and communities over the centuries. Today the fishing industry provides about 17% of animal protein for human consumption and tens of millions of jobs globally, thus playing a key role in food security and human livelihoods (FAO, 2014). In Australia, commercial and recreational fisheries contribute economic and social benefits to the community, with over 5,000 people currently employed in the commercial fishing industry and 3.4 million Australians engaged in recreational fishing each year (Stephan & Hobsbawn, 2014).

While fisheries provide essential sources of food and sustain livelihoods, they inevitably impact the marine ecosystem (Jennings & Kaiser, 1998). Thus the challenge in managing fisheries is to maintain economic and social benefits while achieving acceptable levels of impacts (Jennings *et al.,* 2014). A first requirement for sustainable exploitation is knowledge of the status of ecosystems and stocks. Yet, because today's stock and ecosystem status depends on the extent and magnitude of past impacts, it is essential to understand the impacts of fishing from the beginning of the fishing activity rather than from some time during its development (Jackson & Johnson, 2001; Jackson *et al.,* 2011).

Understanding the causes and extent of impacts from fishing thus requires data that are informative about ecosystem status before and during the different stages of disturbance (e.g. Pauly, 1995; Jackson *et al.,* 2011). For instance, reconstructing the history of resource exploitation uncovers significant ecosystem changes that happened before scientific investigation began (Sáenz–Arroyo *et al.,* 2005; Saenz-Arroyo *et al.,* 2006), and helps to understand the magnitude of these changes (Baum & Myers, 2004; Ferretti *et al.,* 2008). Historical and social analyses also tell of past failures in natural resource policies, thus helping to define today's research and management priorities and identify challenges for future sustainability (Caddy & Cochrane, 2001).

In South East Australia, previous studies that have adopted the historical perspective, and that have made use of old records, have uncovered past ecological changes and have shed light on the practical, theoretical and organizational challenges behind fisheries development and administration. For example, historical data on the sealing industry of Macquarie Island suggested that steep declines in the elephant seal (*Mirounga leonina*) population of the region were evident as early as the 1830s, coinciding with the peak of the sealing industry (Hindell & Burton, 1988). In other studies (Klaer, 2001, 2006), the analysis of catch and effort data from steam trawlers operating along the coast of New South Wales between 1915 and 1960 highlighted substantial decreases in the abundance of some of the species targeted by the fishery (e.g. tiger flathead). Further, a 'then' (1800s) and 'now' (1980s and 2000s) comparison of Tasmanian coastal ichthyofauna found a consistent range reduction for some species and a loss of predatory reef fishes, partly attributed to poor fishing practices (Last *et al.,* 2011). Numerous constraints to fisheries research and management (including logistics and the limitations of available technologies) inevitably limited resource protection in earlier periods. For instance, a review of the administration of Australian fisheries up to 1991 found that one of the major constraint to fisheries development and effective management had been the absence of an integrated national fishery agency that could coordinate fishery research (Harrison, 1991). Also, weak governance of the South East Australia trawl fishery, resulting from incomplete jurisdiction by Federal and State Governments, lead to uncontrolled increases in trawling effort during the 1970s and 1980s (Grieve & Richardson, 2001; Tilzey & Rowling, 2001), and the collapse of commercially important fish stocks, such as the eastern gemfish (*Rexea solandri*) (Smith & Punt, 1997).

Although there are insights into the history of fishing exploitation in South East Australia that provide information on the status of past populations and the regulations in place to manage the resources, these are often limited to a specific time, region, or fishery. Moreover, most of the historical overviews were published more than a decade ago (such as the special issue of *Marine and Freshwater Research,* edited by Smith & Smith, 2001), thus missing the impact of recent exploitation on ecosystem structure. Here I provide an update and comprehensive review of the history of fishing and management in South East Australia, focusing particularly on trawling, and I identify available data that can help inform on long-term changes in fish populations and communities in this region.

The work is divided into four sections. First I explore aboriginal fisheries (before the 1780s), then fisheries of the colonies (late 1700s to early 1900s) along with their impact on the marine ecosystem. Next, I outline the development and management of commercial trawling from 1915 to the present. Last, I discuss the availability of long-term datasets.

2.3 Aboriginal and colonial fisheries

2.3.1 Aboriginal fishing

Australian's Aborigines had long fished the rivers, beaches and estuaries along the coastline of South East Australia. Despite aboriginals first arriving in the region more than 30,000 years ago, early evidence of human interactions with marine environments had been submerged by sea-level rise by the end of the last Ice Age (Pepperell, 2005; Attenbrow, 2010), and first archeological records date back about 1300 to 3000 years. Archeological evidence revealed that marine products, including a wide range of fish (e.g. snapper and yellowfin bream), marine mammals (e.g. seals and whales), shellfish and crustaceans (e.g. oysters and southern rock lobster), made up a consistent portion of the diet of people living near the ocean (Roughley, 1953; Pepperell, 2005; Attenbrow, 2010). The relatively small aboriginal populations exploited marine resources for subsistence, thus most likely impacted the marine environment only in restricted locations and sustainably (Pepperell, 2005; Attenbrow, 2010).

2.3.2 Sealing and whaling

Seal and whale hunting by Europeans were the first large-scale human impact on marine resources in Australia (Thompson, 1893; Gill, 1967; Bach, 1976). Domestic sealing and whaling industries arose soon after British settlers established the colony of Sydney in New South Wales in 1788, and of Hobart in Tasmania in 1804. The primary product was seal and whale oil, which became the first valuable export from the new colonies (Gill, 1967).

The sealing industry had an intense albeit short history. Large colonies of fur (*Arctocephalus pusillus*) and elephant (*M. leonina*) seals were discovered along the Bass Strait shore and islands in 1797 (Gill, 1967; Harrison, 1994). Commercial harvesting soon began and the Bass Strait sealing industry developed rapidly (Harrison, 1994). The success of the industry promoted a rapid search for new resources. In 1810, Macquarie Island with its plentiful seal population was discovered and seal harvesting expanded there (Harrison, 1994). However, by the early 1830s, unregulated harvest of seals and illegal hunting exhausted seal populations of the Bass Strait and Macquarie Island (Murray, 1927; Harrison, 1994; Ling, 2002), collapsing these industries, and finally ending this valuable trade.

The whaling industry contributed to the economy of the colonies for a longer period. Large sperm whales (*Physeter macrocephalus*) were first sighted in 1791 off Maria Island, along the coast of Tasmania, and in close proximity to Port Jackson, and, shortly after, the whaling fishery developed (Thompson, 1893). At first, a poor knowledge of the South East Australian coast and limited capitals to be invested in an offshore whale fishery confined domestic whaling to local bays (Thompson, 1893). The inshore whaling industry saw a rapid expansion during the early 1800s and by the late 1830's more than 40 whaling stations were located around the coast of Tasmania and many others in New South Wales (Harrison, 1994). By the mid 1840s, following intense harvesting, the stocks of small coastal whales were exhausted, and the industry began to decline (Thompson, 1893; Harrison, 1994; Klaer, 2006).

By that time, the improved capital resources of the colonies allowed the colonists to develop an offshore whaling industry. Whale stocks from offshore Australia waters were first harvested in 1798, when English and Spanish whalers were forced out of South American waters and moved to Australia (Thompson, 1893). Sydney and Hobart were established as the main ports for the exploitation of Australian and New Zealand whale populations. In 1830, 22 vessels were sailing out of Port Jackson, mainly targeting sperm and southern right (*Eubalaena australis*) whales (Thompson, 1893). Twenty years later (~1860), the number of whaling vessels operating in the industry had increased fourfold (Thompson, 1893; Bach, 1976; Harrison, 1994; Klaer, 2006).

In the late 19th century, explosive harpoons, steam-driven whaling vessels and the compressor (a mechanism used to pump gas into the whale carcass after death to prevent the whale from sinking) were developed. These new technologies, and the discovery, in 1904, of vast stocks of whales in the Southern Ocean boosted a large scale whaling industry (Clapham & Baker, 2002), which killed approximately two million whales in the Southern Hemisphere between 1904 and 1964 (Clapham & Baker, 2002). Commercial whaling of all species ceased in 1978 in Australian waters (Paterson & Paterson, 1984).

2.3.3 Finfish fisheries

Of less economic importance than sealing and whaling, fishing for finfish was a form of livelihood for the first European settlements (Thompson, 1893; Harrison, 1994; Pepperell, 2005). Although fish was intended to be an important source of fresh food, fishing remained a marginal activity for the first 150 years of settlement (Tenison-Woods, 1882; Tull & Polacheck, 2001; Pepperell, 2005). The limited scale of the fish trade was due to a combination of factors. Among these were the small population size of the Australian colonies, the abundance and accessibility of pastoral and agricultural products, consumer preference oriented towards meat consumption, and greater job opportunities offered by land-based industries, such as agriculture and mining (Dannevig, 1909; Jacobsen, 2010).

In 1880 a New South Wales Royal Commission was formed to survey fisheries and fish stocks. At that time, about 8 line-fishing and 27 seine boats were engaged in fishing the numerous bays of Port Jackson, and regularly supplied the Sydney Fish Market. All main fishing grounds were within a moderate distance of Sydney, and fishing grounds north of Port Stephens and south of Jervis Bay (Fig. 2-1) were so remote that they remained almost untouched, and indeed unknown, by professional fisherman at the turn of the 19th century (Fisheries Inquiry Commission, 1880). The primary target species of line fishing was the snapper (*Pagrus auratus*), considered the most abundant and valuable of all fish. Net fishing included the use of seine, drift and stake nets (nets fixed on the ground and stretched across streams) and was limited to beaches, bays, harbors and river estuaries. Among other common target species were sea mullet (*Mugil cephalus*), yellowfin bream (*Acanthopagrus australis*), whiting (family *Sillaginidae*), black fish (*Girella elevate*), garfish (*Hyporhamphus* *australis*) and flathead (*Platycephalus spp*.) (Fisheries Inquiry Commission, 1880).

The Royal Commission found no sign of overfishing for the key target species, which were reported to be quite abundant along the coast. However, in fishing locations close to Sydney, fishermen reported a consistent decline in fish catches, particularly of the once very abundant snapper. Much of the problem was blamed on the use of nets. To prevent further decline of fish stocks, the *Fishery Act 1881* was introduced, representing the first formal attempt at protecting the natural supply of fish in Australia (Tenison-Woods, 1882; Thompson, 1893).

Following the Royal Commission in New South Wales, in 1882, a Royal Commission into the Fisheries of Tasmania was formed. In 1882 the island's human population was still small (about 115,000 inhabitants; Fenton, 2011), and the Tasmanian economy was dominated by agriculture. Fishing was confined to the estuaries of the Derwent and Tamar Rivers and to near shore waters in the south east of the Island. Among the predominant fishing practices were oyster dredges, lobster traps, seines and handlines. Oysters (*Crassostrea gigas*), lobsters (*Jasus edwardsii*), and striped (*Latris lineata*) and bastard (*Latridopsis forsteri*) trumpeters were the main target species of this fishing industry. In 1882, 100 commercial fishermen and 53 boats were fishing in Tasmania. As fishing was a marginal part of the Tasmanian economy, the Royal Commission surveyed fish stocks when many were in a near virgin state, except for the native oysters beds, already extensively overfished, and the kingfish (now called gemfish - *Rexea solandri*) stock, which was declining in inlet waters (Fisheries Inquiry Commission, 1883; Harrison, 1994).

Overall, whilst some stocks in coastal bays and inlets close to Sydney and Hobart were already heavily exploited, others were almost untouched. Thus fish resources beyond these regions, which included the extensive coast of South East Australia, were largely undeveloped.

In 1901 the Commonwealth of Australia was formed to include the six separate British self-governing colonies. Soon after federation, the Commonwealth Government took its first action into fisheries. In 1906, a sub-committee, appointed by the Federal cabinet, reviewed the work done by the State Fishery Departments on developing fisheries. The work was judged incomplete, and Australian fisheries were considered underdeveloped. The sub-committee also reviewed foreign fisheries and found that countries of Europe, USA, Canada, South Africa and New Zealand had all been successfully fishing with trawlers (Harrison, 1991).

By the early 1900s, trawling – fishing by towing nets across the seabed - had long been the predominant way of catching fish in Europe (Robinson, 1996; Roberts, 2007). When the Australian Federal Government reviewed trawling in Europe, in Great Britain there were more than 10,000 steam trawling vessels, landing about 400,000 t of demersal fish (Thurstan *et al.,* 2010).

Inspired by the success of trawling in other countries, the Australian Federal Government decided to invest in the construction of a trawler that could survey the potential of the continental shelf for trawl fish (Dannevig, 1909; Harrison, 1994). Accordingly, the Australian-built vessel *Endeavour* carried out trawling experiments along the coast of southern Queensland, New South Wales, Victoria and Tasmania between 1909 and 1914. Grounds suitable for trawling were detected and satisfactory fish catches were recorded (Dannevig, 1909).

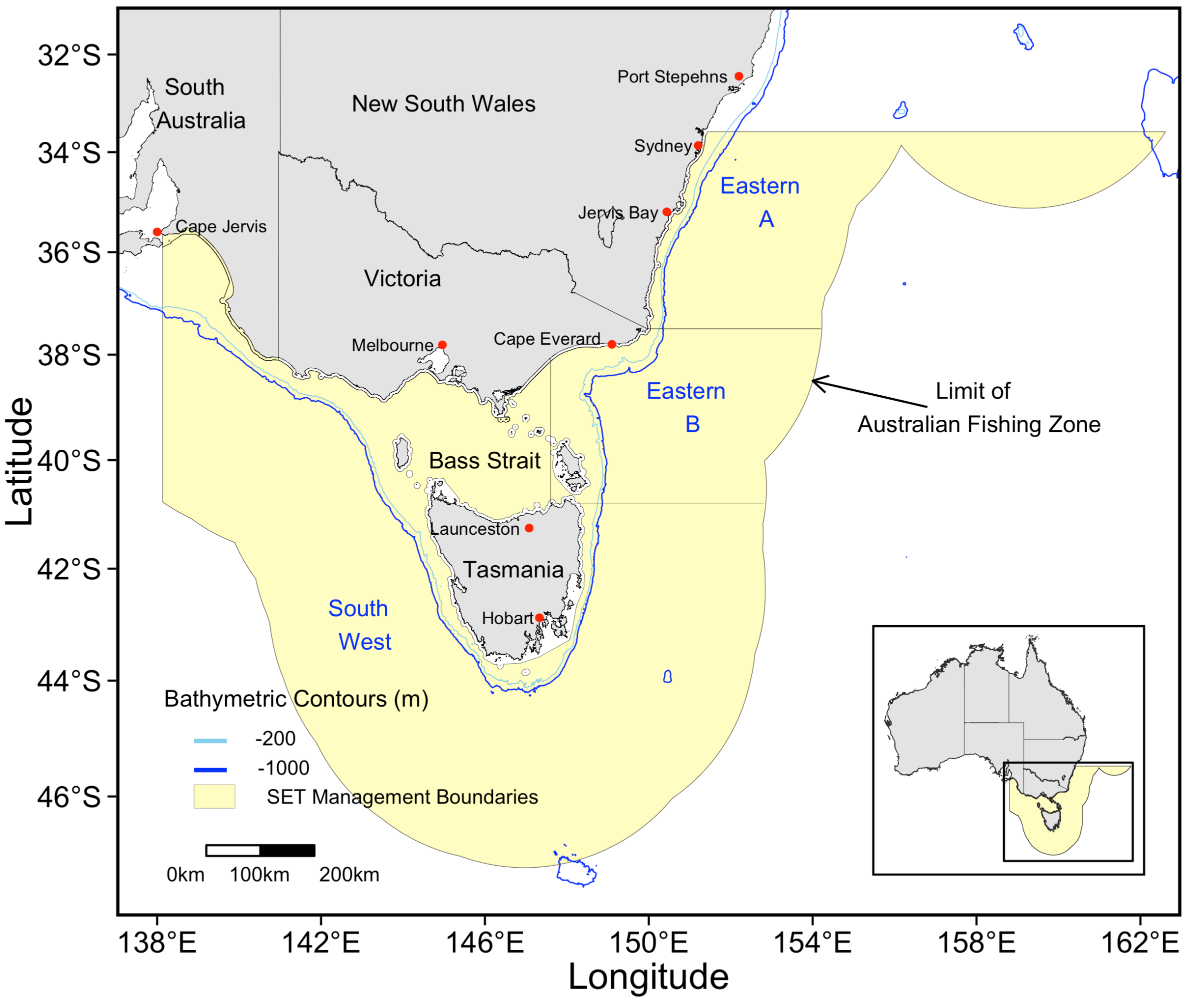

Figure 2-1. Map of South East Australia showing main fishing locations, South East Trawl (SET) fishery management boundaries and sectors.

2.4 Commercial trawling

2.4.1 Steam trawling

Following exploratory fishing, in 1915 the New South Wales government imported three steam trawlers fully crewed with British experienced skippers and crews, and commercial harvesting commenced. This year signaled the beginning of the South East Trawl Fishery (SET), which quickly developed into Australia's major finfish fishery, and since then has remained the primary supplier of fresh fish to the domestic markets of Sydney and Melbourne (e.g. Klaer, 2001; Tilzey & Rowling, 2001).

The SET had a troubled start. In 1920 the New South Wales government built four more steam trawlers, but despite making good catches and meeting a strong demand for the fish, the venture lost £300,000 between 1915 and 1923 (Harrison, 1991). In 1923 the New South Wales Government sold the fleet to private enterprise. By 1929, the private industry owned 17 vessels operating from Sydney and Newcastle targeting mainly tiger flathead (*Platycephalus richardsoni*), latchet (*Pterygotrigla polyommata*) and chinaman leatherjacket (*Nelusetta ayraud*) (Klaer, 2001). As the Australian Army requisitioned men, vessels and fuel during War World II, the fishing fleet steeply declined, and by 1943, only one steam trawler and a few Danish seiners (i.e. the seine net, negatively buoyant, is dragged in a circle around the fish and then hauled from the boat), introduced in 1933, were operating. Soon after WWII the number of steam trawlers stabilized to about 10 (Klaer, 2001; Tilzey & Rowling, 2001), but increasing operational costs and the decline in abundance of flathead in the mid-1950s saw the end of the steam trawling period. Danish seiners, more economical to operate, and able to exploit inshore fishing grounds, replaced steam trawlers, which permanently left the fishery in 1961 (e.g. Klaer, 2001; Tilzey & Rowling, 2001) (Fig. 2-2).

Over the history of steam trawling, fishing grounds had expanded southwards and into deeper waters. It expanded as far as Cape Everard, in Victoria (Fig. 2-1), and the mean fishing depths shifted from 75-100 meters in the early 1900s to 110-130 meters by the 1950s. The catch per unit of effort for retained commercial species declined from more than 200 kg/h in the early 1920s to less than 100 kg/h in the late 1950s. Also the retained catch composition changed over time. Whereas flathead, latchet and chinaman leatherjacket dominated early catches, red fish (*Centroberyx affinis*) and jackass morwong (*Nemadactylus macropterus*) formed the bulk of later catches (Klaer, 2001).

At the time of the SET establishment, fishery legislation in New South Wales was driven by productivity growth and industry development, and based on a longstanding belief that marine resources of the region were plentiful and limitless, thus suitable for large-scale commercial harvesting. Accordingly, the SET opened without any Government restriction on fishing effort. In 1935, following the decline in the abundance of the main target species and recognition of the lack of knowledge about Australia's marine resources, the Commonwealth Government established the Fishery Investigation Section, a CSIR – Commonwealth Scientific and Industrial Research – unit to carry out fisheries and biological research that could guide fisheries' management and development. However, after WWII the need for economic reconstruction and growth shifted the Government focus once again towards industry development, and marine resource management and protection returned to be of little concern (Jacobsen 2010)".

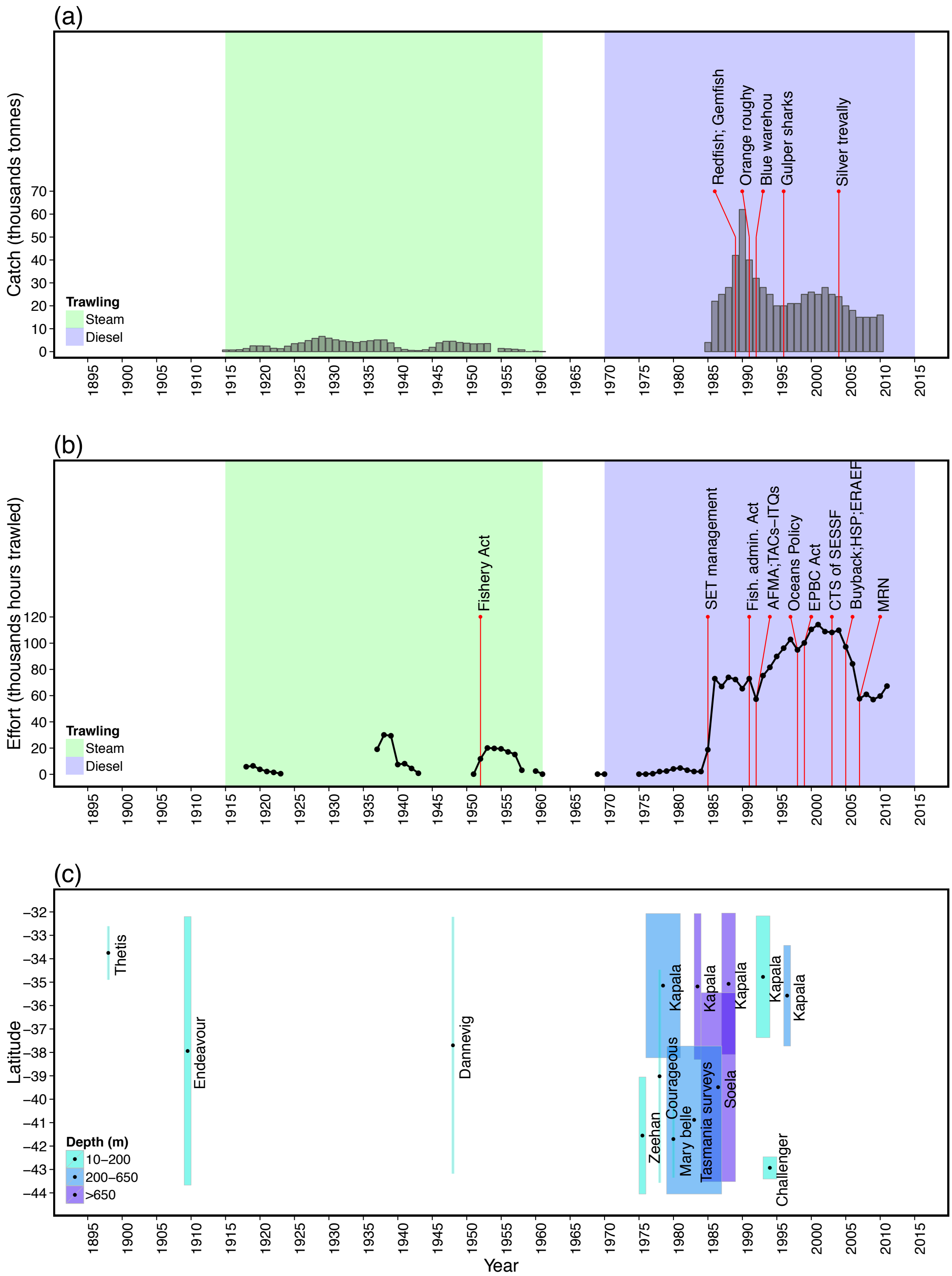

Figure 2-2. Temporal distribution of (a) retained catch and (b) commercial effort of the trawling fleet, for the period 1895-2011 (data from Klaer 2006; Woodhams *et al.* and AFMA), and (c) temporal and geographic distribution of scientific trawl surveys data for the same period. Text refers to (a)

“overfished” stocks, and (b) management actions. In (c) *Tasmania surveys* includes the surveys *Zeehaan 1979*, *Trawl fish resources* and *Challenger miscellaneous*, described in Table 2-2. [1]

[1] Acronyms used in Fig. 2-2 (b): Fish. Admin. Act= Fisheries administration Act; AFMA = Australian Fisheries Management Authority; TACs = Total Allowable Catches; ITQs = Individual Transferable Quotas; EPBC = Environmental Protection and Biodiversity Conservation Act; CTS = Commonwealth Trawling Sector; SESSF = South East Scale and Shark Fishery; HSP = Harvest Strategy Policy; ERAEF = Ecological Risk Assessment for the Effects of Fishing; MRN = Marine Reserves Network.

2.4.2 Diesel powered trawling

In 1971, the first diesel-powered otter trawl was introduced to the SET and this trawling method rapidly expanded throughout the 1970s. Spawning migrations of gemfish (*Rexea solandri*) up the New South Wales coast in slope waters between 300 and 400 m were the main target of the fishery during the winter months. The fishery quickly developed in deeper, upper-slope grounds down to about 600 m, and progressively increased its range southwards to waters around Tasmania and westwards to western Bass Strait. By 1976, gemfish had become the main commercial species in the SET and by 1982 the number of vessels involved in the fishery peaked at 180 (Tilzey & Rowling, 2001).

The rapid expansion of the trawling fleet was helped by a substantial Federal boat-building subsidy and the absence of major management restrictions (Tilzey, 1994; Grieve & Richardson, 2001). In the 1970s the administration of the SET fishery was State-based, with regulations on minimum legal length for certain species, legal minimum cod-end mesh size, and a limit on vessel length to 32 m overall (Tilzey, 1994; Grieve & Richardson, 2001). Despite the enhancement of the *Fishery Act* 1952, which regulates Australian fisheries in waters beyond the 3 miles, the Commonwealth government played a negligible managerial role (Grieve & Richardson, 2001).

The marked increase in the SET fishing fleet during the 1970s was followed by an apparent decline in economic return to operations in the early 1980s (Tilzey, 1994). This resulted in the Commonwealth government considering comprehensive management options for the SET, aimed at securing the long-term profitability of the fishery. Accordingly, in 1985 the Commonwealth government defined the fisheries boundaries, and three management sectors (Eastern Sector A, Eastern Sector B and

South West Sector, Fig. 2-1). Entry criteria were set for each sector and new fishing licenses were permitted only for the South West Sector, still considered under-exploited (Tilzey, 1994). Extra regulations were also introduced. These included a mandatory catch and effort logbook system in 1985, which required operators to report shot-by-shot catch and effort records (Grieve & Richardson, 2001).

In 1986 the discovery of orange roughy (*Hoplostethus atlanticus*) and, soon after, blue grenadier (*Macruronus novaezelandiae*) stocks on the mid- and upper-slopes of Tasmania and Victoria shifted trawling effort from the East to the South West Sector (Tilzey, 1994). Accordingly, Tasmanian landings increased from 4% of the total SET catch in 1979-80 to 37% in 1989, with orange roughy and blue grenadier dominating the catch. The decrease in fishing effort off the New South Wales coast was also stimulated by a decline in the abundance of eastern gemfish (Tilzey & Rowling, 2001).

The collapse of gemfish and orange roughy stocks after a relatively few years of exploitation fomented concerns about the status of the SET fishery and about the effectiveness of the contemporary management regime (Grieve & Richardson, 2001). Management measures for the SET fishery were initially based on input controls (i.e. actions limiting the amount of fishing effort, including when, where and with what gears catches could be taken), but the need to protect individual stocks that continued to decline called for alternative management practices. As such, output controls that focus on the amount of catch that can be harvested were implemented. These included Total Allowable Catch (TAC) limits and Individual Transferable Quota (ITQ) allocations (Flood *et al.,* 2012). Consequently, in 1988, a TAC of 3000 t for gemfish was set and, a year later, an ITQ system was introduced for this species (Woodhams *et al.,* 2011).

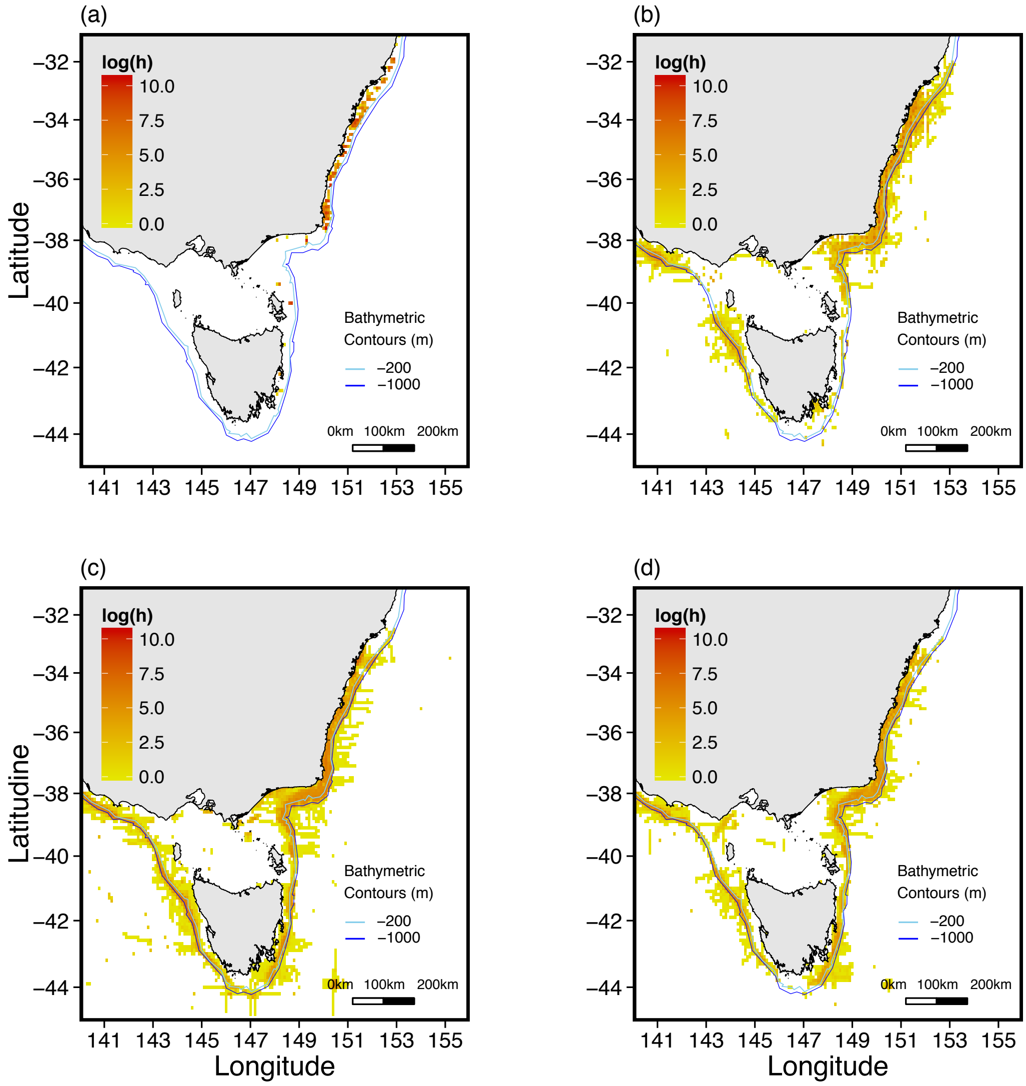

Figure 2-3. Spatial distribution of trawling effort in cumulative (log of) hours trawled in 0.1 degrees cells, in (a) 1915-1960, steam trawling period; (b) 1986, discovery of orange roughy stocks; (c) 2001, peak of trawling effort; and (d) 2007, reduction in trawling effort following management restrictions. Effort data use for this figure is as data used for Fig. 2-2 (b).

Stock assessments became the main scientific tool to assess and help manage SET target stocks. In 1992, TACs and ITQs were introduced for another 15 target species of the SET fishery, including orange roughy, and the Australian Fisheries Management Authority (AFMA) was created under the *Fisheries Administration Act*

1991 (Commonwealth of Australia, 2015)*,* as a statutory authority for the management of fisheries under Commonwealth jurisdiction.

During the 1990s, following the development of a deep-water fishery, the trawling fleet faced a reduction in overall number of vessels (from 180 vessels in 1982 to 108 active vessels in 1997) and an increase in average vessel tonnage and horsepower. This, coupled with the adoption of more sophisticated electronic fishing and navigation systems (e.g. since the mid-1980s, net-sonde and GPS started to be used), resulted in a further increase in the fishing fleet's efficiency (Tilzey & Rowling, 2001). Fishing effort in terms of hours trawled also increased steadily until about 2001 when they peaked at 112,000 hours from approximately 70,000 in 1990 (Figs. 2-1 and 2-2) (Larcombe *et al.,* 2001; Woodhams *et al.,* 2011).

Despite the increase in fishing efficiency and effort, catches steeply declined (Fig. 2-2). The total SET catch peaked at 62,269t in 1990, mainly because of the discovery and exploitation of the orange roughy fishery, and fell to approximately 20,000 t in 1995. Catches never exceeded 30,000 t thereafter (Woodhams *et al.,* 2011). In addition to the collapse of the eastern gemfish stock by 1989, and several orange roughy stocks by the early 1990's, other main targeted species, such as blue warehou and redfish, were also in decline (Tilzey & Rowling, 2001). However, TACs set by fishery managers for some of the major target species either remained at their original levels or increased, suggesting that socio-economic management objectives in some instances prevailed over sustainability aims (Tilzey & Rowling, 2001).

2.4.3 The trawl fishery today

In 2003 the SET fishery was amalgamated into the broader Southern and Eastern

Scalefish and Shark Fishery (SESSF), bringing together three other significant fishing methods on the same or adjacent grounds: the South East non-trawl fisheries, the Southern Shark Fishery (SSF), and the Great Australian Bight Trawl fishery (GABT). Accordingly, the SET fishery became the Commonwealth Trawl Sector (CTS) of the SESSF (Woodhams *et al.,* 2011).

Following 30 years of increased exploitation, the status of many CTS fish stocks had deteriorated. In 2005, five (blue warehou, eastern gemfish, orange roughy, redfish and silver trevally) of 17 species under quota management were classified as "overfished" (stock biomass below the biomass limit reference point, Fig. 2-2), and for another seven species there was inadequate information to make a reliable assessment of the status of the stocks. Additionally, the lack of adoption of formal decision rules for setting TACs to aid stock recovery when biomass limit reference points were reached contributed to further declines in stock biomass (e.g. orange roughy) (McLoughlin, 2006).

Many Australian fisheries were facing similar problems at that time, impoverishment of fish resources and low profitability. For instance, the increase in fuel costs and the increasing value of the Australian dollar relative to currencies of trading partners made Australian fisheries less competitive in international markets and promoted increased import of relatively cheap seafood to meet domestic demand (Rayns, 2007).

This situation raised major concerns about the viability of Australian fisheries, leading to calls for radical changes in management. Changes were also required by a number of policy directions aiming at a wider consideration of the impacts of fishing and other ocean-related activities on all aspects of the natural environment. Among the most significant policies were the National Strategy for Ecologically Sustainable

Development (ESD), released in 1992; the *Fisheries Management Act* 1991; the *Environmental Protection and Biodiversity Conservation Act* 1999 (EPBC); and Australia's Oceans Policy, released in 1998 and integrating ocean governance across sectors and jurisdictions (Smith *et al.,* 2007). All these policies reflected a worldwide trend to ecosystem based fisheries management (EBFM) that saw fisheries management gradually evolving from being primarily focused on a sustainable exploitation of target species to a much wider focus on ecosystems, and the impacts of fishing (Pikitch *et al.,* 2004).

The need to restore Australian, particularly Commonwealth, fisheries and ecosystems resulted in major initiatives (Table 2-1). In 2005, the Securing Our Fishing Future (SOFF) structural adjustment package (Australian National Audit Office, 2015) was announced, and the fishery buyback component of the package resulted in the removal of 50% of the fishing concessions, which was followed by an almost equal reduction in fishing effort (Fig. 2-2). Even without a reduction in quotas, this was predicted to improve the economic viability of the fleet. More importantly, concurrent with the buyback scheme, AFMA implemented both a harvest strategy policy (HSP) (initially implemented as a harvest strategy framework in the SESSF) to set species and stock specific catch limits (Smith *et al.,* 2008), and an ecological risk assessment (ERAEF) to address the broader ecological effects of fishing (Hobday *et al.,* 2011). Most of the quota and main by-catch species are now routinely assessed and recent trends have seen a decline in TACs for most species (Smith *et al.,* 2008, 2013; Woodhams *et al.,* 2011). The ERAEF and subsequent environmental risk management responses resulted in increased spatial management of the fishery with the introduction of area closures and habitat protection (Hobday *et al.,* 2011). In addition, in 2012 the South-east Commonwealth Marine Reserves Network was established, placing major restrictions on commercial fisheries in the SESSF region (Commonwealth of Australia, 2015b). The reserves cover an area of 388,464 km^2 with a depth of 40 - 4600 m and include a range of zonings, from 'sanctuary zones' (i.e. activities limited to not-extractive research) to 'multiple use zones' (i.e. activities limited to those that do not significantly impact benthic habitats or do not have an unacceptable impact on the values of the area). Meanwhile, the whole SESSF management plan was carefully reviewed to identify management options that would lead to better ecological and economic outcomes (Fulton *et al.,* 2014).

2.4.4 Overview of the trawl fishery

Since the introduction of the first trawler in South East Australian waters, there has been dramatic evolution in the industry and its management (Fig. 2-2 and Table 2-1). The fishery grew from a small fishery operating in a confined area of the continental shelf close to Sydney and targeting only a few species to one of the largest Commonwealth fisheries' sectors operating across a broad geographical region and exploiting more than 100 species. During this time, 34 of these species or stocks became quota-managed, and some of them collapsed or experienced steep declines in abundance after intense harvesting (Fig. 2-2). At present, blue grenadier, flathead, pink ling (*Genypterus blacodes*) and silver warehou (*Seriolella punctata*) account for most of the catch and, together with blue-eye trevalla (*Hyperoglyphe antarctica*) and gummy shark, are the most valuable species in the SESSF (Woodhams *et al.,* 2011). Concurrent with the development of the fishery, management restrictions changed from a few regulations on nets and vessel size enforced by States Governments to limited entry, TACs and ITQs systems, area closures and gear restrictions, mostly established by the Australian Commonwealth Government. These changes were driven by shifts in management focus. Initially, lack of knowledge of the effects of fishing resulted in limited restrictions. By the early 1990s, concerns about the stability of the industry in the longer term drove the management focus towards the sustainability of target species. By the late 1990s, recognition of the possible impact of fishing beyond target species promoted an ecosystem-based fisheries management (EBFM) approach (Table 2-1).

Table 2-1. Key management shifts for the South East Trawl fishery.

Year	Key management shifts	Management actions and policies	Main outcomes	Reference
1980s	From States to Commonwealth management	Management plan for the south-eastern trawl fishery (1985)	SET management sectors Entry criteria & fishing licences Mandatory catch and effort logbook system	Tilzey 1994
1990s	From input to output controls	Fisheries Administration & Fisheries Management Acts (1991)	Australian Fisheries Management Authority (AFMA) TACs and ITQs for 16 target species	http://www.comlaw.gov.au
2000s	From sustainability of target species to ecosystem-based fisheries management (EBFM) approach	Securing Our Fishing Future (SOFF) structural adjustment package (2005)	Removal of 50% of the SET (renamed CTS) fishing concessions	http://www.anao.gov.au
		Commonwealth Fisheries Harvest Strategy Policy and Guidelines (HSP) (2007)	Improved species and stock specific catch limits resulting in TACs declines	Smith *et al.,* 2008
		Ecological Risk Assessments (ERAEF) for all Commonwealth fisheries (2005)	Increased spatial management	Hobday *et al.,* 2011
		National Representative System of Marine Protected Areas (NRSMPA) (1998 – commitment)	Australia's South-east Commonwealth Marine Reserves Network (2012 - completed)	http://www.environment.gov.au

2.4.5 Other sources of impacts

During the last century trawling has been the major fishing activity in South East Australia, but not the only one. Traps, lines and other means of net fishing have been and still are important fishing methods. Historically relevant fisheries are the South East non-trawl fisheries, which include hook and line fisheries, and the Southern Shark Fishery (SSF). Hooks and lines were used to catch fish from the period of first European colonization, long before the development of trawling (e.g. Fisheries Inquiry Commission, 1880). At present the Scalefish Hook Sector (ScHS), which uses drop-line and demersal long-line, spatially overlaps and shares most of its target species with the CTS (Woodhams *et al.,* 2011). On the other hand, the shark fishery developed in the 1930s in Victoria and expanded to waters of Tasmania and South Australia due to the high demand for food production and vitamin A from shark liver oil during World War II. The main fishing method was long-lining, replaced by gillnetting from the 1960s onwards, and the main target species was school shark (*Galeorihnus galeus*), later replaced by gummy shark (*Mustelus antarcticus*) due to high content of mercury and steep decrease in abundance of the former (Walker, 1999). Today the Shark Gillnet and Hook Sector (SGHS) is an important sector of the SESSF in terms of landings and economic value (Flood *et al.,* 2012).

In Tasmanian and Bass Strait waters, southern rock lobster (*Jasus edwardsii*), abalone (*Haliotis spp.*) and scallop (*Pectin fumatus)* have all been the basis of valuable fisheries throughout the 20th century (Harrison, 1994). Southern rock lobster and abalone fisheries are still today among the most profitable state-managed fisheries delivering high-quality products for exports (Hartmann *et al.,* 2012; Lyle & Tracey, 2012). In contrast, the once valuable scallop fishery collapsed in the 1980s and is at present restricted to limited areas and seasons (Woodhams *et al.,* 2011).

Some pelagic fisheries were historically important, whereas others arose more recently and are today of substantial economic value. Fishing for 'couta (*Thyrsites atun*) was a predominant activity during the first half of the 1900s in waters off western Victoria and around Tasmania, and ceased around the 1970s (Blackburn & Gartner, 1954; Grant *et al.,* 1978; Bridge, 2009). In contrast, commercial tuna fisheries developed during the 1950s and gained importance since then. Pelagic long-line and purse seine fishing gears are used in these fisheries, and Southern Bluefin tuna (*Thunnus maccoyii*) is the most valuable targeted species, although a range of other species, including sharks, are taken as by-catch (Woodhams *et al.,* 2011).

Also, recreational fishing has been a growing component of the total fisheries harvests in South East Australia, with a rate of fishing participation of about 17%, 12% and 29% of the population in New South Wales, Victoria and Tasmania, respectively. Recreational fisheries target a broad range of invertebrates (e.g. squids, abalone and lobsters) and finfishes (including bony and cartilaginous fishes, e.g. Stevens, 1984; Henry & Lyle, 2003), with flatheads being the primary finfish species group harvested in South East Australia. In 2000 the total annual recreational harvest of finfish was in excess of 27,000 t nationally, compared with the total SET catch of about 30,000 t. This comparison highlights the importance of the recreational catch, which for some species and regions likely equals or exceeds the commercial catch (Henry & Lyle, 2003).

2.5 Long-term datasets

2.5.1 Bottom trawl surveys

Interest on the status of wild fish stocks in South East Australia began with two Royal Commissions, one in New South Wales (1880) and the other in Tasmania (1882) (Fisheries Inquiry Commission, 1880, 1883). These reported the first decline in the abundance of coastal fish stocks close to the main commercial ports.

Shortly afterwards, and at other times throughout the history of trawling, several trawling experiments and surveys were carried out in the region. Some of these studies precede the beginning of the SET (i.e. the beginning of commercial trawling), whereas others assessed the demersal resources at different stages following commencement of exploitation. The early trawling experiments aimed at developing demersal fisheries, i.e. finding grounds suitable for trawling and fish resources to be exploited commercially (e.g. Farnell & Waite, 1898; Dannevig, 1909).

The earliest small-scale bottom trawl trials were undertaken during the late 1880s and the 1890s along the coast of Victoria and New South Wales and were organised by private enterprises, but none resulted in satisfactory catches (Dannevig, 1909) (Table 2-2).

In 1898, The New South Wales Government launched the first bottom trawl survey along the New South Wales coast, undertaken by the vessel *Thetis*. The survey lasted for five weeks and proved successful in locating trawlable grounds along the 180 miles of coast explored. In support of the development of a trawling industry in New South Wales, Captain Nelson, who led the trawling operations and was an experienced North Sea fisherman, noted that the amount of edible fish caught during the survey was similar to that of comparative studies undertaken in the North Sea (Farnell & Waite, 1898). However, details on fish quantities and fish sizes were not reported, and the findings were of limited use for the development of a commercial fishery (Dannevig, 1909).

Following federation in 1901, the Australian Federal Government funded a more extensive bottom trawl survey program in 1909 (Harrison, 1991), which aimed at discovering marketable fish species along the South East Australia coast and determining their quantities. Capitan Dannevig, born and trained in Norway and regarded as leading fisheries expert in Europe, was appointed Commonwealth Director of Fisheries (1908) and took charge of the acquisition of the vessel, named *F.I.S. Endeavour* (Fig. 2-3), and the supervision of scientific investigations. The *F.I.S. Endeavour* carried out trawling experiments along the coast of South East Australia between 1909 and 1914 (Dannevig, 1909). After operating for 5 years, the vessel and the crew of 23 men were lost in 1914 in waters close to Macquarie Island (Harrison, 1991). The loss of the *F.I.S. Endeavour* and the high cost of the project discouraged the Federal Government from replacing the *F.I.S. Endeavour* with a new research trawler to investigate demersal fisheries (Harrison, 1991).

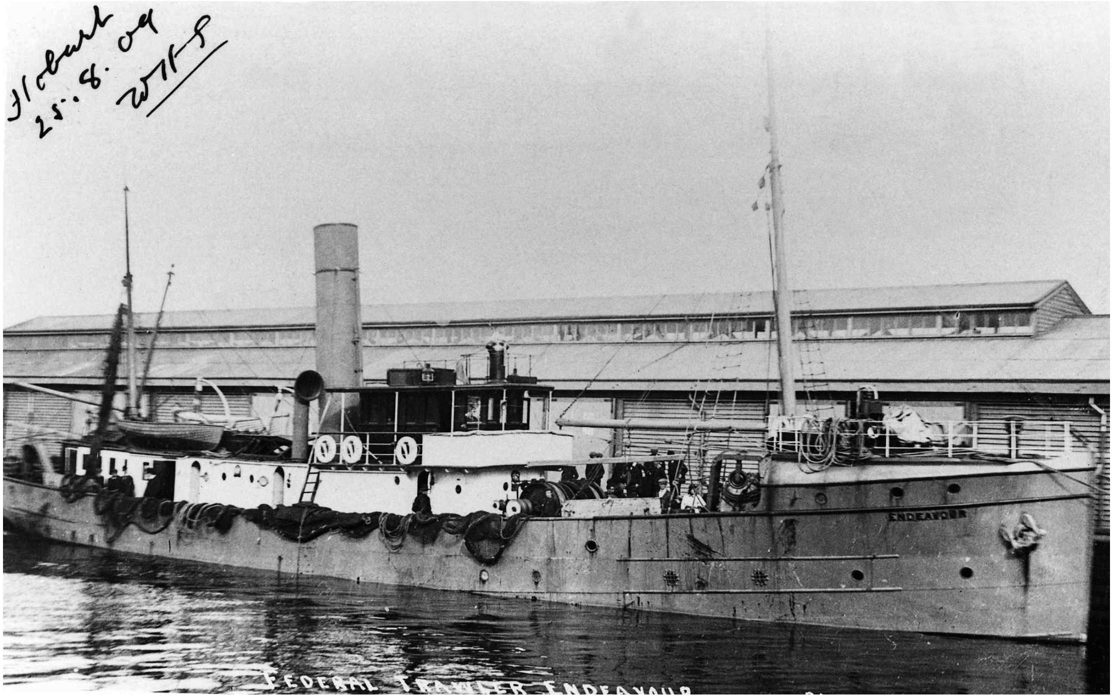

Figure 2-4. *Endeavour* research vessel.

Following the development of the trawl fishery in New South Wales in 1915 and during the steam-trawling era (1915-1961), the focus of demersal fisheries research centred on the main target species (particularly tiger flathead, jackass morwong and redfish) (Tilzey & Rowling, 2001). During this time, fieldwork on demersal fisheries was limited and only a couple of scientific trawling expeditions were organised by State and Commonwealth fisheries agencies (Mawson *et al.,* 1988; Harrison, 1994).

Between 1966 and the early 1970s, Japanese research vessels explored the Australian coast. Among these were the *Oshuru Maru*, which carried out trawling surveys from Fremantle, in Western Australia, to Sydney, in New South Wales, and the *Umitaka Maru* and *Kaiyo Maru*, which undertook a small number of tows in waters around Tasmania (Last & Harris, 1981). At about the same time, the Australian *F.R.V. Urania* surveyed the South, East and North Tasmanian coasts (Webb & Wolfe, 1977). However, records from most of these surveys remained unpublished. I was able to locate scattered records, reporting partial catch and effort data, for the surveys *Oshuru Maru, Umitaka Maru* and *Kaiyo Maru* in CSIRO archives, but I could not find records for the *Urania*.

When diesel-powered trawlers were introduced in South East Australia in the early 1970s, the SET fleet rapidly expanded and searched for new trawling grounds. Accordingly, fisheries research agencies organised a considerable amount of exploratory fishing and trawl surveys during the 70s and 80s. The main aims were to identify new trawl grounds and to evaluate the commercial potential of target species (e.g. Lyle, 1993). This was particularly the case for the continental shelf of Tasmania, which had limited exploration until the late 1970s, and for all continental slopes of South East Australia.

Most of the surveys were state-based. In Tasmanian waters, the *F.V. Zeehaan* and the *F.R.V. Challenger* carried out trawling trials at a range of depths between 1975 and 1979, and 1978 and 1987, respectively. Also, in the early 1980s the Tasmanian Fisheries Development Authority conduced a major trawl survey program off southern Australia using multiple vessels (Lyle, 1993; Koslow *et al.,* 1994; Tilzey & Rowling, 2001). Further north, in Victorian waters, the vessels *Ray Larsson*, *Margaret Goulden* and *San Antone* carried out pair bottom trawling between 1973 and 1975 (Webb & Wolfe, 1977), and a couple of years later the vessel *Battle Axe* carried out trawling surveys along the central Victorian coast (Last & Harris, 1981). In the mid-1980s an extensive 4 year survey of the slope of eastern Bass Strait by the *F.R.V. Sarda* and CSIRO's *F.R.V. Soela* was completed (Wankowski & Moulton, 1986; Tilzey & Rowling, 2001), and a similar survey of western Bass Strait was later conducted in 1987–90, using chartered commercial vessels (Tilzey & Rowling, 2001).

Meanwhile, The *F.R.V. Kapala* extensively surveyed the demersal resources of New South Wales during the 1970s and 1980s (these surveys are documented in a serie of *Kapala* reports, e.g. Gorman & Graham, 1978, 1983; Graham, 1990) (Table 2-2).

In addition to State-based research programs, the CSIRO Division of Fisheries Research carried out demersal trawl cruises and surveys in SET waters. During the late 1970s the *F.R.V. Courageous* carried out several trawl cruises along the continental shelf of Tasmania and southern New South Wales, despite the vessel being primarily employed for the investigation of jack mackerel (*Trachurus declivis*) and other benthopelagic species (e.g. Brown *et al.,* 1978). Additionally, between 1987 and 1989 the *F.R.V. Soela* carried out trawling surveys on the continental slopes around Tasmania, with a major focus on orange roughy distribution and biology (Koslow *et al.,* 1994).

In the 1990s the SET management regime shifted towards output controls. The introduction of a compulsory, shot-by-shot logbook system for the trawl and Danish-seine fleets in 1985 provided catch-and-effort data for stock assessment and the setting of TACs. However, data on fish size and age composition were generally lacking. As these data were needed for the models used to assess stock status, in 1994 the Integrated Scientific Monitoring Program (ISMP), an onboard and port-based monitoring program, was established to provide essential information for selected species (Knuckey & Gason, 2001). In addition, several scientific surveys were conducted in the 1990s, including a continuation of surveys by the *Kapala 1992-94*, which assessed fish stocks on the continental shelf of New South Wales (Graham *et al.,* 1996; Chen *et al.,* 1997), and by the *F.R.V. Challenger 1993-95*, which extensively surveyed the southern and eastern continental shelf of Tasmania (Jordan, 1997). Also, in 1996-97 surveys that replicated these undertaken by the *Kapala* between 1976 and 1977 were undertaken to study changes in relative abundance and size composition of commercial fishes and sharks on the continental slope of New South Wales after 20 years of trawling (Andrew *et al.,* 1997; Graham *et al.,* 2001). This was the most recent of the surveys carried out in the region until the establishment of a SET-wide fishery independent survey in 2010 (Knuckey *et al.,* 2013).

In summary, a range of bottom trawl surveys and trawling cruises were carried out in South East Australia (considering waters off New South Wales, Tasmania and Victoria, Fig. 2-1) during the last 100 years that covered the period of trawling exploitation. A list of all surveys identified in the literature is provided in table 2-2. Latitudinal range and temporal coverage characterising bottom trawl surveys for which I collected catch and effort data are shown in Fig. 2-2 (c). (See chapter 3 for details on data collection and standardisation).

Table 2-2. Bottom trawl surveys carried out in South East Australia. (*) Catch and effort data collected in this study. Codes are as follows: N=New South Wales, V=Victoria, T=Tasmania, SH=shelf, SL=slope, USL=upper slope and MSL=mid-slope.

Surveys and vessels	Years	Region	Depth	Source
Private enterprises	1888	N	SH	Klaer, 2006
Lady Lock	1889	V	SH	Dannevig, 1909
Otter, Dory & Charlotte Fenwick	1891	V	SH	Dannevig, 1909
Thetis *	1898	N	SH	Farnell & Waite, 1898
Endeavour *	1909-10	T, V, N	SH	Dannevig, 1909
Liawanee	1944	T	SH	Harrison, 1994
Dannevig *	1948	T, N	SH	Mawson *et al.,* 1988
Oshuru Maru	1966	V, N	SH	Last & Harris, 1981
Umitaka Maru & Kaiyo Maru	1967	T	SH	Last & Harris, 1981
Urania	1969-70	T	SH	Webb & Wolfe, 1977
Ray Larsson & San Antone	1973-75	V	SH	Webb & Wolfe, 1977
Zeehaan *	1975-76	T	SH	Webb & Wolfe, 1977
Kapala *	1976-77	N	USL	Graham *et al.,* 1997
Zeehaan & Craigmin	1977	T, V	SH	Last & Harris, 1981
Battle Axe	1977	V	SH	Last & Harris, 1981
Kapala *	1977-78	N	USL	Gorman & Graham, 1978
Courageous *	1978	T, V	SH	Brown *et al.,* 1978
Zeehaan *	1979	T	SL	Last & Harris, 1981
Kapala *	1979-81	N	USL	Graham *et al.,* 1997
Challenger miscellaneous *	1979-87	T	SH, SL	Lyle at al., 1993
Mary Belle *	1980	T	SH	Lyle at al., 1993
Trawl fish resources phase 1 *	1981-82	T	SL	Lyle at al., 1993
Trawl fish resources phase 2-3 *	1982-82	T	SL	Lyle at al., 1993
Trawl fish resources phase 4 *	1983-83	T, V	SL	Lyle at al., 1993
Kapala *	1983-84	N	MSL	Gorman & Graham, 1983
Sarda & Soela	1984-89	V	SL	Wankowski & Moulton, 1986
Soela *	1987-89	T, N	SL	Koslow *et al.,* 1994
Kapala *	1987-89	N	MSL	Graham, 1990
Chartered commercial vessels	1987-90	V	SL	Tilzey & Rowling, 2001
Kapala *	1993-94	N	SH	Chen *et al.,* 1997
Challenger *	1993-95	T	SH	Jordan, 1997
Kapala *	1996-97	N	USL	Andrew *et al.,* 1997

2.5.2 Other sources of data

Scientific expeditions beside bottom trawl surveys were carried out during the early developmental stage of commercial fishing in South East Australia. These include a series of areal observations carried out intermittently between 1936 and 1946 by Stanley Fowler from the CSIRO Division of Fisheries, who successfully looked for large schools of pelagic fish along the Tasmanian coast. This collection of about 13,000 nitrate-based negatives, including aerial photos and a historical documentation of the pelagic fishing industry, is now stored in CSIRO archives in Canberra. Among the other scientific expeditions were the long-line surveys undertaken to determine whether sharks could be caught in commercial quantities in Southern Australia. A first survey was carried out in Tasmania in 1942, with the vessel *Aralla*. Then, in 1948 the vessel *Liawanee*, chartered by CSIRO, surveyed the coasts of Tasmania, Victoria and Southern New South Wales for sharks and 'couta' (*Thyrsites atun*), though with unsatisfactory results. Again, between 1950 and 1955, the vessel *Derwent Hunter* set long-lines along the same region to catch sharks (logbooks in paper format can be found in the CSIRO archives in Hobart for the surveys by *Aralla* and *Derwent Hunter* and in OHTA archives in Hobart for the survey by *Liawanee*). However, these data are often limited to very few surveys/cruises and for some surveys, information on the catch is particularly patchy.

Industry related data represent an alternative source of long-term datasets. In South East Australia, logbook data from commercial trawling have been used to assess changes in population abundances since the early stages of bottom trawling exploitation, although data are only available for a limited number of commercial species (Klaer, 2001, 2004, 2006).

Datasets from recreational fishing, such as spear fishing competitions, may also represent a valuable source of information. Some of these data have been collected resulting in a dataset spanning about 50 years (1961-2010) and covering the coast of New South Wales and Victoria. Analysis of this dataset revealed a climate-change related shift in species distribution (Gledhill *et al.,* 2013).

2.6 Discussion

This review identifies aspects of the history of fishing in South East Australia together with the information available since the 1880s to understand the impacts of fishing on marine communities over that time-span. The arrival of European settlers in the late 18th century signaled the beginning of more intense exploitation of marine resources in this region, with unregulated sealing, whaling, oyster dredging and line fishing for snapper resulting in the first significant impact on marine populations and communities, and all causing declines in the abundances of the stocks targeted. Exploitation extended to demersal resources when a trawling industry developed off the coast of South East Australia in 1915. The fishery was initially limited to the continental shelf of New South Wales, but during the 1970s it expanded to southern and deeper waters. Inadequate regulations, and the tardy management response to early evidences of declines in targeted fish stocks were the main cause of a number of stock collapses between the mid 1980s and 1990s (Fig. 2-1). Fishing methods other than trawling have also impacted both demersal and pelagic ecosystems of South East Australia. One example is the southern shark fishery, partially responsible for a major decline in school shark abundance.

I assessed the availability of retrospective records and long-term datasets that may be informative about past ecological changes, and identified data from scientific bottom trawl surveys carried out between 1898 and 1997. All these surveys were planned to explore well-defined depth regions (continental shelf or upper slope or mid slope) and geographical areas (mainly New South Wales, Tasmania and Bass Strait), and for none of the combinations of depth, region and geographical area there is information on the variety and abundance of species (or other taxonomic units) at frequent or regular points in time over the entire past 100 years. Additionally, the datasets identified share most of the complications common to other long-term datasets (Magurran *et al.,* 2010). Sampling equipment, sampling design, effort, spatial extent, taxonomic resolution and scientific objectives varied from study to study as the result of changes in research priorities (e.g. major focus on assessing the demersal resources or major focus towards single species such as orange roughy, for example), capital investment and skill and technological improvements. For example, changes in sampling gear and navigation equipment may have influenced the detectability of species or individuals across surveys. Also, species identification skills and taxonomic resolution increased with time leading to finer classification, thus higher number of species recorded in more recent surveys. Furthermore, as species abundances were reported either as number of individuals or biomass, or occasionally were not recorded at all, comparing abundance indices over this time-span will prove challenging. All these factors complicate community comparisons in time and impact the ability to draw robust conclusions. (See chapter 3 for details on the information reported in each survey, data quality and standardization).

Despite these complications, the information available does cover the entire history of commercial fishing and should therefore provide insights into the impacts of fishing on south-eastern Australian ecosystems. Further, for more than one region, survey data are available on demersal fish communities before the beginning of commercial exploitation. This is, for example, the case of the continental shelf of New South Wales and Tasmania, surveyed in 1898 and 1909, respectively, and that of the continental slope of New South Wales, surveyed before and after 20 years of trawling (in 1976 and 1996, specifically). While some of the surveys carried out after the 1970s have been previously analyzed, this review brings together all the current knowledge that has been recorded for the entire period since the late 1880s. However, the diversity and detail in the methods of data collection and the types of data collected is likely to require new methods of analysis. As this diversity and detail is common in developing fisheries globally (e.g. Ferretti *et al.,* 2008), exploration of methods to analyse such historical data is needed if we are to understand better the historical impacts of fishing.

The next chapters of this thesis will explore methods for interrogation and analysis of the South East Australian data.

3 Chapter 3 – Bottom trawl survey data collection and standardization

3.1 Introduction

In chapter 2 I identified bottom trawl surveys carried out in South East Australia between 1898 and 1997, thus sampling demersal communities of this region at different times through the history of commercial trawling. These surveys were performed by various research agencies, which collected and organized catch and effort data in different formats; also the detail of the information reported changed across surveys and over the years. Despite the value of these data, there has not yet been a systematic effort to collect, digitalize and standardize all of the information available. Therefore, I compiled catch and effort data from bottom trawl surveys identified in chapter 2 into a single dataset. This chapter describes the processing carried out and the assumptions made to convert the data to a format suitable for analysis, and summarizes some important aspects of the resulting dataset.

3.2 Methods

3.2.1 Data collection and digitalization

I searched public and private archives, and libraries to locate survey reports in paper format, as well as publications on survey findings and information that would help the standardization of historical data (e.g. regional fish guides that provide information on past common and scientific names). Public archives searched included the CSIRO historical archive and the Tasmanian Archives Heritage Office (TAHO), in Hobart. Private archives searched were Anthony Harrison's collection on the history of fishing in Tasmania, stored in LINC Tasmania, Rosny Park, and Neil Klaer's collections on the history of trawling in South East Australia. Libraries searched were the CSIRO library in Hobart, the State Library of Tasmania (http://www.linc.tas.gov.au), the Biodiversity Heritage Library (http://www.biodiversitylibrary.org) and the Open Library (http://openlibrary.org). Having identified these sources, I digitalized catch and effort data from the historical surveys for which I was able to retrieve the corresponding reports (i.e. *Thetis* 1898, *Endeavour* 1909 and *Dannevig* 1948).

Next, I examined databases to collect survey data that had already been digitalized. Databases searched were the New South Wales Department of Primary Industries (NSW DPI) database; the Institute for Marine and Antarctic Studies in Tasmania (IMAS) database; and the Commonwealth Scientific and Industrial Research Organization (CSIRO) database.

I was able to assemble catch and effort data for a total of twenty bottom trawl surveys carried out between 1898 and 1997. These surveys are listed in Table 2-2.

3.2.2 Data standardization

Tow position (i.e. latitude and longitude) and net characteristics (i.e. headrope length and cod-end mesh size) were missing for some surveys, and taxonomic resolution changed over time along with the scientific names of some species. To fill these data gaps I calculated tow positions and I made educated assumptions about net characteristics. These assumptions are required because net characteristics influence net selectivity (i.e. the net's ability to catch a certain size or kind of fish) and therefore need to be considered when data collected using different sampling gears are compared (Reeves *et al.,* 1992; Maunder & Punt, 2004). Also, the headrope length is essential to calculate the area swept in each tow, which is commonly used as a measure of sampling effort and was calculated for each survey's tows. Next, to obtain species lists comparable across surveys I updated species names and adopted a common species coding system. Lastly, I converted all survey catch and effort data to a common format. Each step of the data standardisation is detailed in the following sections.

3.2.2.1 Tow positions

I calculated survey tow positions in latitudinal and longitudinal degrees when this information was lacking. Tow positions for the *Thetis*, *Endeavour* and *Dannevig* surveys, here defined as *historical surveys*, were reported using landmarks (e.g. Port Stephen) and depth. Using the approach detailed in Klaer (2006), I constructed a table containing landmark positions in latitudinal and longitudinal degrees, and the positions where a straight line from each landmark crosses the 200 m and 1000 m contours. For each tow I assigned a position in latitude and longitude according to landmark position and a linear interpolation of tow depth. (See Table 8-1 in Appendix 1 for Landmark positions, and positions at 200 m and 1000 m used for the conversion).

3.2.2.2 Net assumptions

Net mesh sizes (cod-end) were not available for the *historical surveys* and for most of the surveys collected from the IMAS database and therefore had to be assumed. Due to the commercial focus of these surveys (these were exploratory surveys designed to help develop a commercial fishery), I relied on specifications of commercial trawl nets used at the time of the survey. I assumed a net mesh size of 3 inches (76 mm) for the *historical surveys* (Fairbridge, 1948), and 90 mm for IMAS surveys (Jeremy Lyle, personal communication). For the latter, if the target species was orange roughy, I assumed a mesh size of 110 mm (Jeremy Lyle, personal communication).

Likewise, net headrope lengths were not reported for the *Thetis*, the *Mary Belle* and for tows undertaken by the vessel *Bluefin* during the *Trawl fish resource survey*. For the *Thetis* I assumed a headrope length of 21 m, as reported for the *Dannevig*, which surveyed similar grounds. I concluded that *Thetis* and *Endeavour* nets did not have the same headrope length despite these surveys being undertaken in closer years. This was because the larger *Endeavour* net (with a headrope length of 29 m) was adopted in Australia for the first time in 1909, after the *Thetis* survey (Dannevig, 1909). The *Mary Belle* was an inshore survey and its net was most likely relatively small. Hence, I assumed the same headrope value as reported for the *Challenger* 1993-95 (26 m), which also explored inshore Tasmanian waters (Lyle, 1993; Jordan, 1997). Lastly, I assumed headrope lengths of 40 m for the *Bluefin* tows because this is the length of a standard multispecies commercial net operating on the outer continental shelf, where these tows where carried out (Lyle, 1993).

3.2.2.3 Sampling effort

Depending on the survey, sampling effort for each tow was given either as time trawled, distance covered or swept area. I calculated trawling effort in terms of area swept in km^2 per tow. I estimated swept area following Sparre & Venema (1989):

$$A = D * hr * X2 \qquad (3\text{-}1)$$

Where *D* is the distance covered (also given as $D=V*t$, where *V* is the trawling speed and *t* is the time trawled); *hr* is the headrope length; *X2* is that fraction of the headrope length, *hr*, which is equal to the width of the path swept by the trawl, the 'wing spread', and its suggested value is 0.5. When the trawling speed was not reported I assumed it to be 5.6 km/h (~3 knots) because this is the standard trawling speed reported for the majority of tows in both the *historical* and more recent surveys (Dannevig, 1909; Jordan, 1997).

3.2.2.4 Species names

I crosschecked for species names no longer in use. For the *historical surveys* species names were given as species common names used at the time the survey was carried out. To obtain a species list comparable across surveys, I interpreted and translated species common names into scientific names using a combination of lists of biological records collected during some of these surveys and provided by the Australian Museum, in Sydney, survey reports (e.g. Farnell & Waite, 1898), and available literature on past species taxonomic classification (e.g. Tenison-Woods, 1882; Ogilby, 1886; Stead, 1906; Roughley *et al.,* 1916; Roughley, 1953).

For many records, the species common name referred to a family or a group of species (e.g. flatheads, sharks). In most of these cases scientific names are at family or higher taxonomic levels. However, in some instances I was able to confidently assume the species belonging to the family reported. For example, I assumed that 'Tasmanian silver belly' referred to *Parequula melbournensis* because this is the only species belonging to the *Gerreidae* (silver belly) family known to inhabit Tasmanian waters (Atlas of Living Australia, 2015). Further, I assumed that 'Tasmanian flounder' referred to *Rhombosolea tapirina* because this is the most common flounder found in Tasmania and was caught in large quantities during the survey in which it was reported (John Pogonoski, personal communication).

Moreover, some old common names may have been linked to a range of current scientific names. For example, among the most uncertain common names were cods and perches. According to the old literature, cods could have referred to either the family *Moridae* or the family *Scorpaenidae*, whereas perches could have referred to any one of the families *Serranidae*, *Sebastidae*, *Neosebastidae* and *Callanthiidae*. When records were uncertain or details from a survey biological record were missing I consulted fish taxonomists at CSIRO, who advised on the most likely species/family. This advice took into consideration the size of the catch, and the depth and specific locality (i.e. latitude and longitude) of capture. Depth and latitude of capture also allowed correcting for some misreported names. For example, the *Endeavour* registered sawfish (*Pristidae*) catches at latitude of about 40° S, but this family is found almost exclusively in tropical waters (Atlas of Living Australia, 2015). Instead, saw sharks (*Pristophoridae*) inhabit temperature waters and are commonly found around Tasmania (Atlas of Living Australia, 2015), so I changed sawfish records (at latitude of 40° S) into saw sharks. (See Table 8-2 in Appendix 1 for species names used in *historical survey* reports and corresponding old and current scientific names).

Also, for all surveys, I updated species names following classifications reported in the Codes for Australian Aquatic Biota (CAAB) (Yearsley *et al.,* 1997), and for each species/family I assigned the relevant CAAB code. As most of the survey catch data consisted of demersal bony fishes and elasmobranchs, I considered only these taxa.

Finally, I cleaned the dataset by removing tows reported inland and tows missing latitude, longitude or depth, as well as tows with no catch information (e.g. the *Thetis* survey reported a few tows with null catches). Also, for all surveys I converted tow latitude and longitude to decimal degrees and depth to meters, if given otherwise, and I calculated tows' mean latitude, longitude and depth, when starting and ending values were provided. I adopted a common format for all surveys, detailed in Table 3-1.

Data manipulation carried out in this study and described above was implemented using the software R 3.1.0 (R Development Core Team, 2014).

Table 3-1. Dataset format and fields explanation.

Field name	Specification	Type
database	databse of origin	character
survey	survey name	character
tow_ID	tow-unique ID	character
day	day the tow was carried out	numeric
month	month the tow was carried out	numeric
year	year the tow was carried out	numeric
season	season the tow was carried out	character
lat	tow mean latitude in decimal degrees	numeric
long	tow mean longitude in decimal degrees	numeric
depth	tow mean depth in m	numeric
net_ID	net-unique ID	character
net_design	net characteristics	character
headrope_m	length of the net's headrope in mm	numeric
codend_mm	size of the net's mesh at the cod-end	numeric
mouth_mm	size of the net mesh at the mouth	numeric
trawling_speed_km	towing speed during the tow in km	numeric
time_trawled_h	time trawled during the tow in hours	numeric
distance_trawled_km	distance trawled during the tow in km	numeric
swept_area_kmq	area swept during the tow in km squared	numeric
vessel_ID	vessel-unique ID	character
vessel_t	vessel tonnage	character
vessel_m	vessel length in m	character
vessel_type	vessel characteristics	character
echosounder	echosounder type	character
radar	radar type	character
other_equipment	other equipment for navigation	character
kapala_report	Kapala cruise report number	character
CAAB	species CAAB code	numeric
species	species scientific name	character
family	species family name	character
class	species class name	character
counts	species counts per tow	numeric
weight	species weight per tow	numeric

3.3 Results

The result of applying the rules and standards outlined above is a dataset containing a total of 3,083 tows taken at depths between 9 m and 1280 m. These tows sampled a total of 574 species belonging to 194 families among chondrichthyes and osteichthyes. The position of survey tows and their temporal distribution are shown in Fig. 3-1 and the survey characteristics and outcomes are summarised in the next paragraphs.

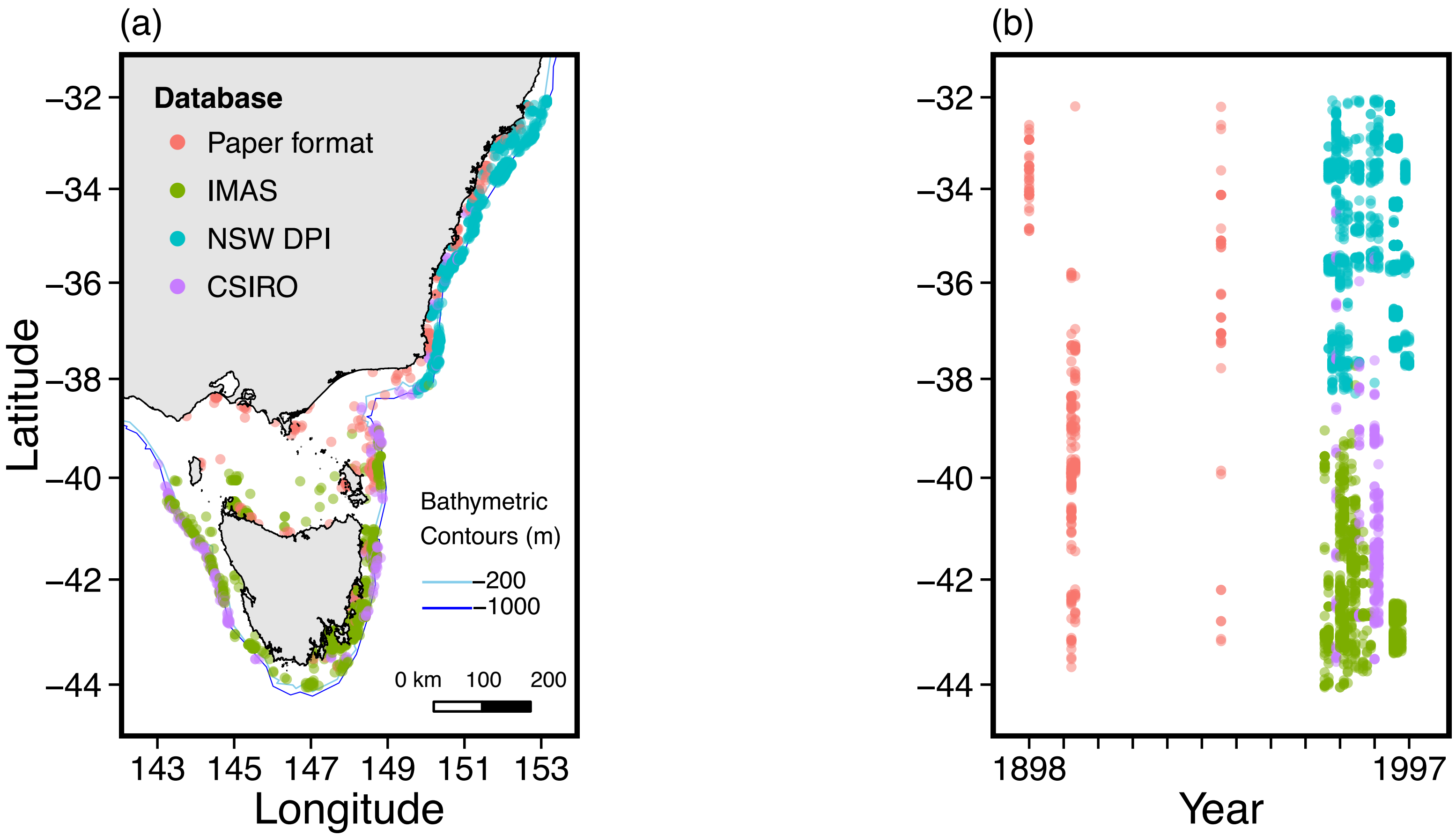

Figure 3-1. Distribution of data: (a) surveys' tows locations; and (b) Latitudinal (in bins of 0.1 DD) and temporal coverage of surveys' tows.

3.3.1 Early surveys

I retrieved data from the *historical surveys* from archives and personal collections. In particular, Neil Klaer kindly provided *Thetis* and *Endeavour* reports, while I found the *Dannevig* survey report (Fig. 3-2) in the CSIRO historical archives, in Hobart. Although the *Endeavour* operated for 5 years (1909-1914), reports of its surveys that I was able to locate only referred to the years 1909 and 1910. It is most likely that the data collected in later years went missing when the vessel and all the crew were lost at sea in 1914. Details on vessel tonnage and length, as well as measures of the trawl net used to sample, were available for the *Endeavour* and the *Dannevig* surveys, but not for the *Thetis* survey (Table 3-2).

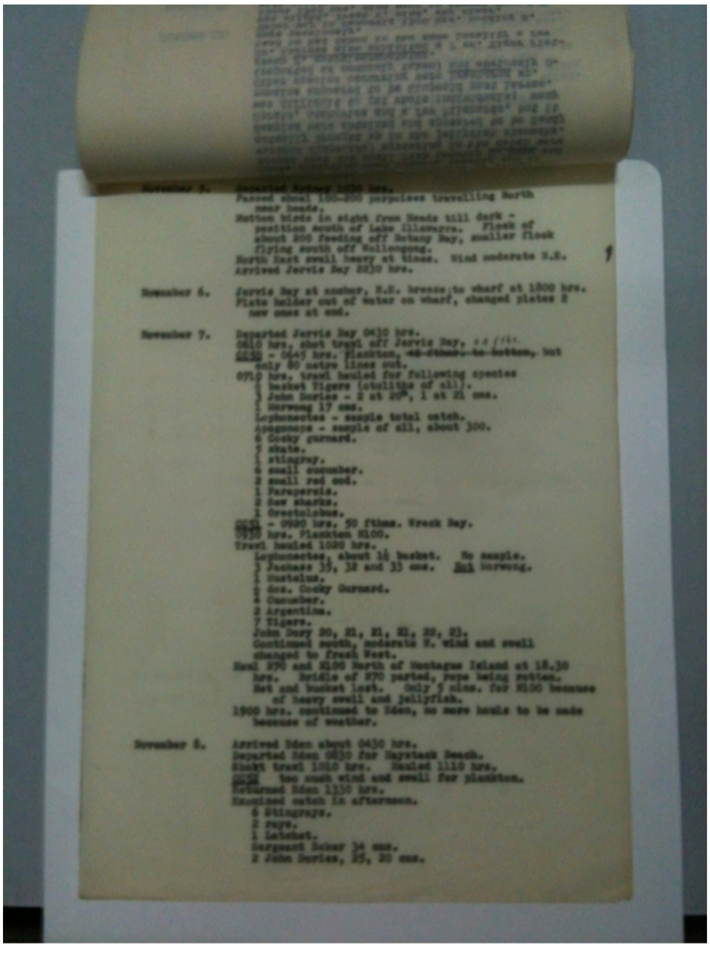

Figure 3-2. *Dannevig* survey report.

Table 3-2. Nets and vessels used in *historical surveys* and their specifications. (*) Assumed values.

Survey	Vessel (t)	Vessel (m)	Vessel type	Net ID	Headrope (m)	Cod-end (mm)
Thetis			New South Wales Government's research vessel	THE	*21	*76
Endeavour	335	41	Australian Federal Government's research vessel	END	29	*76
Dannevig	92	22.6	CSIRO research vessel	DA2	21	*76
				DA1	24	*76

Summaries of the total catch per survey show that the *Thetis* recorded the highest number of species and families, despite having the lowest number of tows (Table 3-3). However, the reported number of species and families depends on the accuracy of taxonomic classification, and, for these surveys, high percentages of the catch records were reported at higher taxonomic level than species (e.g. *Pristiophorus spp*.; "*Seriolella brama & Seriolella punctata"*) or even family (e.g. "*Platycephalidae* – undifferentiated"; sharks). As this percentage was consistent for the *Endeavour* (64% for species and 8% for families), I would expect that the number of species and families sampled during this survey was much higher than reported. This is particularly the case for sharks and rays, which had no distinction of species or family. Information on species abundance was also often lacking. Whereas the *Endeavour* surveys report numbers of individuals sampled for all catch records, this information is available for only 27% of the *Dannevig* catch records and for none of the *Thetis* records.

Table 3-3. *Historical surveys* data summary and quality.

Survey	Tows	Species	Families	Individuals	Taxon > species (%)	Taxon > family (%)	Records reporting individuals (%)
Thetis	43	46	62		54	2	0
Endeavour	218	34	43	244275	64	8	100
Dannevig	47	26	42	755	52	7	27

3.3.2 NSW DPI surveys

The NSW DPI database includes a collection of bottom trawl surveys carried out between 1976 and 1997, with the New South Wales Division of Fisheries research vessel *Kapala* (e.g. Gorman & Graham, 1978, 1983; Graham *et al.,* 1997). All surveys were performed with the same vessel, but nets used changed across surveys (Table 3-4).

Table 3-4. Nets used in NSW DPI survey and their specifications.

Survey	Net ID	Net design	Headrope (m)	Cod-end (mm)
Kapala 1976-77	F6	Boris box, 30 m bridles & 45 m sweeps	21	90
Kapala 1977-78	F6	Boris box, 30 m bridles & 45 m sweeps	21	90
Kapala 1979-81	F3	Engel balloon, 54 m bridles & 45 m sweeps	56	90
	F6	Boris box, 30 m bridles & 45 m sweeps	21	90
Kapala 1983-84	F3	Engel balloon, 54 m bridles & 45 m sweeps	56	42
	F6	Boris box, 30 m bridles & 45 m sweeps	21	90
Kapala 1987-89	F5	Boris box, 30 m bridles & 50 m sweeps	30	90
				42
Kapala 1992-94	F1	Engel balloon, 53 m bridles & 180 m sweeps	56	42
Kapala 1996-97	F6	Boris box, 30 m bridles & 45 m sweeps	21	90

Summaries of the total catch per survey in number of species, families and individuals sampled are given in Table 3-5. Overall, low percentages of catch records were reported at higher taxonomic level than species and no catch record was reported at higher taxonomic level than family. Species abundance is consistently reported as number of individuals sampled per species across all surveys, and no information on species weight is given. For some of these surveys (i.e. *Kapala 1976-77, Kapala*

1996-97 and Kapala 1993-94) length frequency data are also available, although not collected as part of this study.

Table 3-5. NSW DPI survey data summary and quality.

Survey	Tows	Species	Families	Individuals	Taxon > species (%)	Taxon > family (%)	Records reporting individuals (%)
Kapala 1976-77	233	143	85	145583	8	0	100
Kapala 1977-78	58	168	93	31023	2	0	100
Kapala 1979-81	197	134	76	180302	8	0	99
Kapala 1983-84	94	143	53	34156	3	0	100
Kapala 1987-89	165	178	62	68500	3	0	100
Kapala 1992-94	620	256	104	3189381	1	0	95
Kapala 1996-97	165	145	74	130547	4	0	100

3.3.3 IMAS surveys

The IMAS database includes a collection of bottom trawl surveys carried out by the Division of Sea Fisheries (formerly Tasmanian Fisheries Development Authority and later Department of Sea Fisheries) between 1975 and 1995. A combination of research and fishing vessels and nets were used (Table 3-6).

Table 3-6. Nets and vessels used in the Division of Sea Fisheries surveys and their specifications. (*) Assumed values.

Survey	Years	Vessel ID	Vessel (t)	Vessel (m)	Vessel type	Net ID	Net design	Headrope (m)	Cod-end (mm)
Zeehaan	1975-1976	*Zeehaan*	92	22.3	Fisheries vessel	F&B	Frank and Bryce, 45 m bridles & 270 m sweeps	36.3	*90
Zeehaan	1979	*Zeehaan*				URI		38	*90
						CAQ	Coastal Aquarius	47	*90
						C3B	Coastal 3 bridled	36.6	*90
						CBO	Coastal box trawl	17.3	*90
						G3B	Grundy 480 3 bridled	22	*90
Challenger miscellaneous	1979-1987	*Challenger*	87	21	Tasmania Fisheries Department research vessel	F&B	Frank and Bryce, 45 m bridles & 270 m sweeps	29	*90
						URI		38	*90
						GRU	Bridport Grundy 3 bridled	24	*90
						URI2		41	*90
						COMM	Standard commercial	40	*90
						RWA	Roughy net	34	*110
						SAM	Sammy's net	29	*90
						CAQ	Coastal Aquarius	47	*90

Table 3-6. Continued.

Survey	Years	Vessel ID	Vessel (t)	Vessel (m)	Vessel type	Net ID	Net design	Headrope (m)	Cod-end (mm)
Mary Belle	1980-1980	*Mary Belle*	29	14.6	Fisheries vessel	WTR	Frank and Bryce wing trawl	*26	*90
						NZF	New Zealand flounder net	*26	*90
						OTH	Other small trawl nets	*26	*90
Trawl fish resources phase 1	1981-1982	*Challenger*				URI		38	*90
		Bluefin	387	34	Maritime College research vessel	ENG	Engel balloon trawl	*40	*90
Trawl fish resources phase 2-3	1982-1982	*Bluefin*				ENG	Engel balloon trawl	*40	*90
		Challenger				RWA	Roughy net	34	*110
		Petuna Endeavour	200	24	Fisheries vessel	ITL	Italian	60	*90
Trawl fish resources phase 4	1983-1983	*Petuna Endeavour*				ITL	Italian	60	*90
		Challenger				URI		38	*90
		Margaret Philippa	200	26	Fisheries vessel	EBT	Engel high lift balloon trawl	53	*90
						BTA	Baltar trawl with bobbin gear	56	*90
Challenger	1993-1995	*Challenger*				AJ	Otter trawl net, 25 m bridles & 38 m sweeps	26	20 liner

Summaries of the total catch per survey in number of species, families and individuals sampled or species weights are given in Table 3-7. The number of species and families sampled are particularly low for the *Zeehaan 1975-76* and the *Mary Belle 1980* (both coastal), despite the high number of tows. The percentage of catch records reported at higher taxonomic level than species is as high as 10-15% for some surveys, although in none of the surveys were catch records reported at higher taxonomic level than family. Information on species abundance was given as either species weight (i.e. *Zeehaan 1975-76* and the *Mary Belle 1980*), or a combination of species weight and number of individuals sampled (e.g. *Trawl fish resources surveys*). The *Challenger 1993-94* is the only survey in the whole dataset consistently reporting both species weight and number of individual samples for each tow.

Table 3-7. IMAS survey data summary and quality.

Survey	Tows	Species	Families	Individuals	Weight (kg)	Taxon > species (%)	Taxon > family (%)	Records reporting individuals (%)	Records reporting weights (%)	Records with no biomass information (%)
Zeehaan 1975-76	154	14	11	2925	50823	0	0	0	100	0
Zeehaan 1979	43	105	68	351	39535	10	0	37	64	0
Challenger miscellaneous	214	147	78	35173	38138	6	0	81	23	1
Mary Belle	152	18	18	0	3409	0	0	0	100	0
Trawl fish resources phase 1	25	90	54	1342	11393	10	0	27	73	0
Trawl fish resources survey 2-3	65	89	50	8943	70788	13	0	40	60	0
Trawl fish resources phase 4	57	101	58	10576	113553	15	0	74	26	1
Challenger	240	114	68	124945	29429	2	0	97	99	0

3.3.4 CSIRO surveys

The CSIRO database includes a collection of surveys carried out with the *F.R.V. Soela* and *Courageous,* during the 1970s and the 1980s. Both vessels used similar nets (Table 3-8).

Summaries of the total catch per survey in number of species, families and individuals sampled or species weights are given in Table 3-9. Data quality for the *Soela* survey is lower than for the other surveys carried out at similar times. For instance, 19% of the catch records are reported at higher taxonomic level than species, 7% are reported at higher taxonomic level than family, and 38% lack information on species abundance. Species abundance was given as a combination of number of individuals sampled or species weight in both *Soela* and *Courageous* surveys.

Table 3-8. Nets and vessels used in CSIRO survey and their specifications.

Survey	Vessel type	Net ID	Net design	Headrope (m)	Cod-end (mm)
Courageous	CSIRO research vessel	CS3	Frank & Bryce, 228 mm mesh size at wings, 40 mm cod-end liner, Karmoy doors	25.6	40
		CS2	Frank & Bryce, 228 mm mesh size at wings, Fearnought doors	25.6	40
Soela	CSIRO research vessel	CS1	Engel demersal high lift	35.3	37 liner
		CS4	Frank & Bryce, 228 mm mesh size at wings, 40 mm cod-end liner, Polyvalent doors	25.6	40

Table 3-9. CSIRO survey data summary and quality.

Survey	Tows	Species	Families	Individuals	Weight (kg)	Taxon > species (%)	Taxon > family (%)	Records reporting individuals (%)	Records reporting weights (%)	Records with no biomass information (%)
Courageous	50	91	58	2217	10744	3	2	63	53	6
Soela	240	132	81	56409	72212	19	7	34	59	38

3.4 Discussion

I compiled catch and effort data from twenty bottom trawl surveys sampling demersal fish communities of South East Australia between 1898 and 1997 into a single dataset ready for analysis. Whereas data from surveys performed after the 1970s have been analyzed before, data from historical surveys (i.e. before the 1950s) remain largely unexplored. Also, the information retrieved has not been analyzed as a whole before.

Catch and effort data from *Endeavour* and *Thetis* is of particular value because these were the only data collected before the development of a trawl fishery in Australia (chapter 2). Whereas the strength of the *Endeavour* is that it reports abundances for all catch records, thus providing information on community structure (i.e. species/families relative abundances) before exploitation, the strength of the *Thetis* is that it provides a more detailed list of species sampled, thus delivering information on community composition (i.e. species presence).

NSW DPI provided the set of surveys with the greatest sampling resolution. These surveys sampled the demersal communities of the continental shelf and slope of New South Wales between 1976 and 1997. Almost all catch records were reported at species level and the number of individuals sampled per species was given for all survey tows. Because the NSW DPI dataset includes surveys carried out at different stages of commercial exploitation and in some cases prior to fishing (i.e. the *Kapala 1976-77*, which surveyed the continental slope) these data have been used in several previous studies aiming at a better understanding of the effect of trawling on demersal communities and fish stocks (Andrew *et al.,* 1997; Graham *et al.,* 2001; Tuck, 2011; Foster *et al.,* 2015).

All steps of the data standardization process highlighted the marked gaps in data quality between *historical* and the more recent surveys. For instance, for *historical surveys* I had to assume a number of pieces of essential information, such as tow position, net characteristics and species names. In some cases, the information I relied on was imprecise, and may have biased my assumptions. For examples, tow landmarks used to calculate tow positions were sometimes vague (e.g. landmark positions reported as 'Between Haystack Bay and North end of Twofold Bay'), as were some of the species common names reported in survey logbooks (e.g. perch). However, when considering historical data there is almost always a tradeoff between accuracy and having any data to consider at all. Instead, the focus should be on extracting the best information available and accounting for data gaps and limits when interpreting outcomes. Because the challenges I faced are likely common to the standardization of other historical datasets, I hope that my approach can be used in similar contexts.

Data gathering and standardization is an important step in all studies involving data analysis, but this step requires additional effort and time when dealing with historical data. First, available data have to be identified through an extensive literature review. Next, it is necessary to determine where the data may be stored. This can be straightforward if the data are already in digital format (as were data from the 1970s onwards) and more complicated if the data are still in paper format (and buried in archives). In such cases, once records are retrieved they need to be digitalized. Next, data gaps and incongruences have to be assessed and assumptions need to be made so that the information is comparable across different sources of data (e.g. all records needed tow position and area swept). For some data (i.e. that from *historical surveys*) consequential assumptions need to be made, thus deserving thoughtful consideration.

This involves further search across the literature for details that cannot be found in particular survey reports (e.g. fish common and scientific names and details on the fishing nets used at the time surveys were carried out). However, all efforts are worthwhile if the outcome is a long-term dataset that, despite limitations (e.g. coarse taxonomy or basic recording), may be informative about the historical impact of fishing on marine communities, and the meticulous approach detailed here will most likely lead to the finding of such valuable data, perhaps available in many regions around the world.

4 Chapter 4 - Historical baselines and long-term changes to demersal fish communities of South East Australia

4.1 Abstract

Natural communities have long been impacted by human activities, but quantification of human-driven changes often relies on recent data so that knowledge of the full extent of these changes is missing. Marine communities of South East Australia have a shorter and less intense history of exploitation than many comparable temperate marine systems around the world, providing a rare opportunity to understand ecological changes since the beginning of exploitation. The demersal fish communities of the continental shelves of South East Australia were surveyed before the establishment of a trawl fishery in 1915, and lightly exploited prior to that. Here, I analyze bottom trawl survey data from the late 19th century and compare the results to their modern counterparts. My aim is to establish an historical baseline for South East Australian demersal fish communities and key commercial fishes, and to quantify long-term changes over the history of fishing. I used nominal catch rates and frequency of occurrence in surveys spanning over a century to examine changes in all components of the fish community, and GLM models to obtain standardized indices of abundance before and after commercial exploitation for three targeted families (gurnards, flatheads and morwongs). I found marked changes in some components of the demersal fish communities and steep declines in many key commercial fishes, most likely related to the effect of fishing.

4.2 Introduction

Although humans have exploited marine resources since early occupation of coastal areas around the world (Jackson *et al.,* 2001; Richter *et al.,* 2008), harvest greatly intensified during the last 150 years, which saw the industrialization of fisheries (Roberts, 2007) and the consequent marked impoverishment of marine resources and communities (Jackson *et al.,* 2001; Jukic-Peladic *et al.,* 2001; Roberts, 2007; Ferretti *et al.,* 2008, 2010). Worldwide, the development of fisheries followed a common pattern characterized by a first phase of unregulated harvest, a second phase of rising concerns about the sustainability of marine resources, and a third phase of more intense scientific research aimed at understanding fishing-induced changes to marine populations and aiding fisheries' management. However, because of considerable time lags between these phases, resource assessments of changes to fish stocks and communities are often based on data that were first collected at the later stages of exploitation, and there is often no information on the full extent of impacts on fish stocks and communities, nor of their natural (pre-fishing) state (Pauly, 1995; Baum & Myers, 2004). This lack of baseline knowledge and the scientific community's shortsighted view of the effect of fishing on fish stocks and ecosystems may result in misleading management and recovery targets because management and recovery of depleted stocks and ecosystems may be constrained by the magnitude of previous (unquantified) declines (Myers & Worm, 2003).

In South East Australia, the relatively recent development of the commercial fishing industry, its initial low intensity, and the availability of fishery-independent data on nearly pristine communities give us the rare opportunity to assess the full extent of ecological changes due to fishing. Here European colonisation began about 200 years ago and, although seal and whale stocks were soon heavily exploited, fishing for finfish remained a small-scale family business through the 19th century. It was confined to shallow waters close to the main ports of Sydney and Hobart, and beach seines and line fishing were the most common means of catching fish (Thompson, 1893; Harrison, 1994; Pepperell, 2005). It was not until 1915 that a trawl fishery developed, and up to 1961 the fishery remained confined to the continental shelf of New South Wales and involved a small number of vessels (a maximum of 17, in 1929). In the mid-1970s the fishery increased its range to deeper (continental slope) and southern (Bass Strait and Tasmanian) waters, and between the mid-1980s and the 2000s trawling effort increased (Larcombe *et al.,* 2001; Tilzey & Rowling, 2001; Klaer, 2004). Although trawling expanded to include most available South East Australian grounds, it remained marginal on the continental shelf of Tasmania. Here traps, seines, hooks and lines and gillnets had been the main fishing methods and their use increased through the 20th century (Harrison, 1994; Bridge, 2009). One example is the development of a long-line (later gillnet) shark fishery, which began in Victoria about the 1930s and soon broadened its range to include Tasmania and South Australia (Walker, 1999).

Scientific bottom trawl surveys, aimed at evaluating the potential of the trawl industry in the region, were carried out before the development of trawling on continental shelves and also preceding its later expansion onto the slopes, in both cases sampling communities either lightly exploited or untouched by fishing. For instance, the research vessel *Thetis* surveyed grounds off central New South Wales in 1898, and the vessel *Endeavour* surveyed the continental shelf around Tasmania and Bass Strait in 1909 (Farnell & Waite, 1898; Dannevig, 1909). The expansion of the trawl fishery onto the continental slope during the 1970s coincided with the beginning of intense fishery research that documented the development of this industry and its impact on deeper resources (Gorman & Graham, 1983; Tilzey & Rowling, 2001).

Whereas data from the 1970s onwards are commonly used to assess the impact of fishing on fish stocks (Tuck, 2011) and communities (Savina *et al.,* 2013; Foster *et al.,* 2015), data from the *Thetis* and *Endeavour* surveys remained unused, so that important changes in demersal fish communities of the region may have been missed.

Here I analyze data from these early surveys and compare the results to their modern counterparts. My aim is to establish an historical baseline for New South Wales and Tasmanian demersal fish communities and key commercial fishes, and to quantify long-term changes.

When available, historical data are often underutilized (as is the case of the Australian dataset here considered) partly because data format may restrain their accessibility and immediate use (e.g. data may still be in paper format or hidden in archives), and partly due to challenges that are implicit in the analysis of historical datasets (Magurran *et al.,* 2010; McClenachan *et al.,* 2012, 2015). For example, the information reported in surveys older than a century may differ greatly from that of recent surveys because of changes in details (e.g. taxonomic resolution) and methods of data collection (e.g. sampling equipment and technological improvements). Such differences can complicate comparisons over time and affect the ability to draw robust conclusions. This study provides approaches to analysis that overcome some of these limitations, so that broad patterns of community change can be detected.

4.3 Methods

I considered two regions along the South East Australian coast and data from the first and the last survey/s exploring demersal communities. These were 1) the continental shelf of central New South Wales, surveyed by the *Thetis* in 1898, and by the Integrated Scientific Monitoring Program (ISMP) from 1994 onwards (ISMP data available for this study covered the years 1994-2011); and 2) the continental shelf of Tasmania and Bass Strait, surveyed by the *Endeavour* in 1909-1910, and by the *Challenger* between 1978 and 1987 (Figs. 4-1 and 4-2). I referred to the former set of surveys (*Thetis* and *Endeavour*) as representing the *before* period and to the latter (ISMP and *Challenger*) as representing the *after* period.

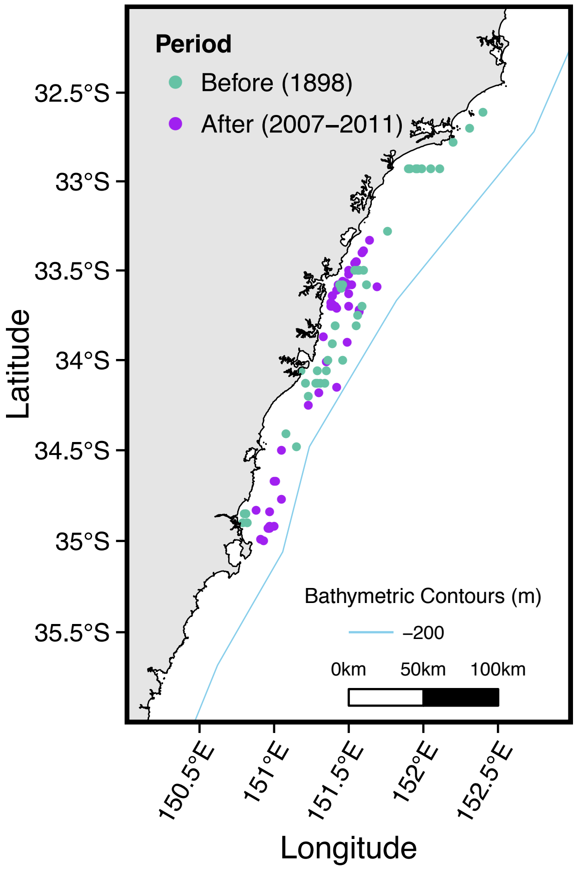

Figure 4-1. Tow positions for region 1.

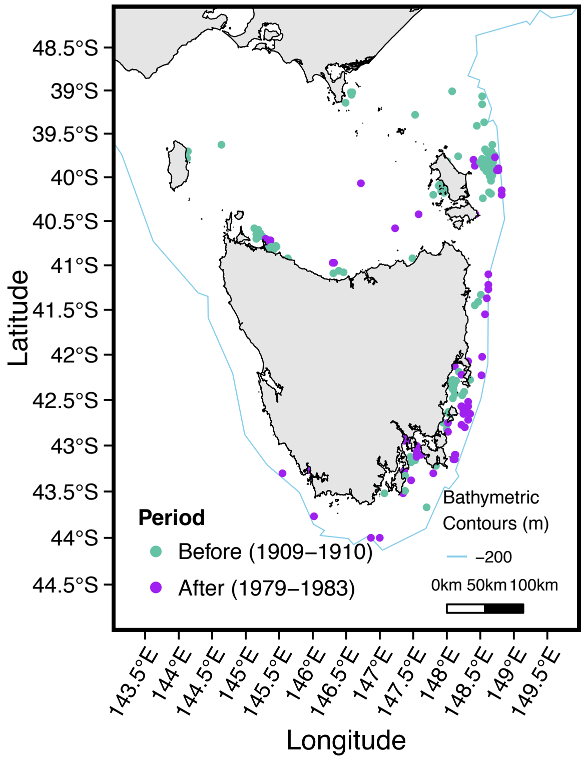

Figure 4-2. Tow positions for region 2.

To explore changes in all components of the demersal fish communities, for each region I compared indices of families' presence or abundance between periods. Also, for region 2, where biomass data were available, I used Generalized Linear Models (GLMs), which accommodate unbalanced sampling, to obtain standardized indices of abundance before and after commercial exploitation for three targeted families (gurnards, flatheads and morwongs).

4.3.1 Data standardization

For each region, I standardized the sampling across time periods.

First, I defined the regions' spatial boundaries. The area covered by the *Thetis* and the *Endeavour* delimited the spatial boundaries of region 1 and 2, respectively. For region 2 I considered latitudes south of 38.5 ° because, in both periods, most of the tows available were carried out in this area. Then, I selected ISMP tows within the spatial boundaries of region 1, and *Challenger* tows within the spatial boundaries of region 2 (Figs. 4-1 and 4-2).

Next, I choose the most comparable set of tows between periods. In particular, for the *after* period (both regions) I considered only tows carried out with a cod-end mesh size of 90 mm, as this mesh size was the most similar to the one used during the *before* period (76 mm, both regions). Also, for region 1 I excluded ISMP data collected before 2007 because in early years the program aimed to monitor stocks of commercial species, and so catch data for non-commercial species may not have been reported consistently in this early time.

Further, I standardized the catch data according to differences in taxonomic resolution between periods. Catch data for the *Thetis* and *Endeavour* was mostly limited to families or groups of families sampled in each tow (e.g. for the *Endeavour* all shark and rays families were aggregated into broad *shark* and *rays* groups). In contrast, surveys for the *after* period reported the scientific name of all species sampled in each tow, with few records at family level. To standardize the information across surveys, for region 1 I lumped families according to the classification used in the *Thetis* (e.g. lizardfishes and relatives ["*Chlorophthalmidae* & *Paraulopidae* & *Bathysauroididae* & *Bathysauropsidae*"], conger eels ["*Congridae* & *Colocongridae*"], and stingarees ["*Urolophidae* & *Plesiobatidae"*]) and for region 2 according to that used in the *Endeavour* (e.g. sharks, rays, flounders and soles [including "*Pleuronectidae* &

Soleidae & *Bothidae* & *Paralichthyidae"*], and dories [including "*Cyttidae* & *Zeidae"*]). I used families and family groups as the taxonomic unit of my analysis. Furthermore, as this study focused on demersal communities, I excluded pelagic families (which were occasionally caught by the demersal nets).

Last, I standardized families' presence or biomass information between periods. For instance, the only information available for the *Thetis* survey was species/family presence-absence, so, for region 1 I considered only presence-absence data for both survey periods. For region 2, indices of abundance were available for all surveys, although in different forms. The number of individuals for each family/family group sampled was available for the *Endeavour*. A combination of number of individuals or weight for each species sampled was available for the *Challenger*. To standardize the information across surveys, from the *Challenger* I excluded tows that did not consistently report species-specific numbers of individual.

The resulting dataset for region 1 consisted of catch data from 43 tows carried out during the *Thetis*, and catch data from 54 tows from the ISMP. For region 2 I had catch data from 152 tows carried out during the *Endeavour*, and catch data from 68 tows carried out during the *Challenger* (90% of these tows were carried out in 1979-80).

4.3.2 Indices of presence and abundance

To explore ecological changes in communities sampled during the periods considered, for region 1 I calculated the frequency of occurrence of each family sampled in each period. This was the sum of tows in which the family was sampled in a specific period (*before* or *after*) divided by the total number of tows carried out in that period, and expressed in percentages. Similarly, for region 2 I calculated the nominal catch rate of each family sampled. This was the sum of the number of individuals sampled (total catch) in a specific period divided by the area swept (total effort) in that period and expressed as number of individual per km^2 (N/km^2).

For each region, I accounted for the different number of tows per period. First I determined which period had fewer tows. For region 1 this was the *before* period, with 43 tows, whereas for region 2 this was the *after* period, with 68 tows. Then I randomly selected (without replacement) an equal number of tows from the remaining period (i.e. 43 tows from the *after* period in region 1 and 68 tows from the *before* period in region 2). Using these data I calculated the frequency of occurrence or the nominal catch rate of each family. I repeated the process 100 times. Then, for the region/period combinations for which I used a subset of tows I averaged the resulting indices of presence or abundance, and calculated the standard deviation. Lastly, I compared the obtained frequency of occurrence and catch rates across periods and looked for marked changes in these indices. Additionally, to determine whether the two indices (frequencies of occurrence and catch rates) gave consistent results, for region 2 (where both indices could be calculated) I compared families' frequencies of occurrence with families' catch rates.

4.3.3 Categories of change

I assigned a category of change to each family sampled. These categories consider the change status of families using the *before* survey data as a baseline. I had a category 1 of change if families were not present in the *after* period (i.e. disappeared); category 2 if families exhibited a decrease from *before* to *after*; category 3 if they increased in frequency of occurrence or catch rate; and category 4 if they were present only in the *after* period. For families of categories 2 and 3 I calculate the percentage of change in either frequency of occurrence or catch rate between periods.

4.3.4 Ecological features

To better understand the changes at family level I considered which species belonging to each family were sampled, and characterized each of these species according to ecological features. As information at species level was available only for the recent surveys (*after*) and it was very patchy for old surveys (*before*), I mainly based my considerations on the species pool representing a family in the recent surveys. I used a combination of data from Fishbase (Froese & Pauly, 2015, extract of species ecology matrix on March 2015) and data collected during the ERAEF project (Ecological Risk Assessment for the Effects of Fishing, Hobday *et al.,* 2011) to characterize each species for their trophic level, maximum body size (in length), resilience to fishing pressure (high, medium, low and very low), latitudinal distribution (tropical, subtropical, temperate and other), and water column distribution or association with particular habitats (demersal, benthopelagic and reef-associated). I used resilience to fishing pressure categories as specified in Fishbase, which are based on (Musick, 1999), and calculated using values of selected life-history parameters (i.e. intrinsic rate of population increase, growth coefficient, fecundity, life span and age at first maturity).

4.3.5 Key families

I selected sets of families based on:

(1) The most frequently encountered or the most abundant families for each period, to determine shifts in main family catch composition between the two periods. For

region 1 these were families with a frequency of occurrence of at least 50% (recorded in at least 50% of tows), whereas for region 2 these were the most abundant families, which cumulatively accounted for 50% of the total catch.

(2) Less influential families, in terms of total catches, characterized by marked changes between the two periods. For region 1 these were decreasing (category 2) or increasing (category 3) families, with frequency of occurrence between 20% and 50% in one of the two periods and characterized by a change in occurrence between the two periods of at least 50%. Similarly, for region 2 I selected families of categories 2 and 3, with catch rates greater than 50 N/km^2 in one of the two periods (but excluding families already considered as the most abundant ones), and characterized by a change between the two periods of at least 50%. The 50% threshold was set to consider clear changes, less likely related to differences in sampling between periods (e.g. mesh size).

(3) Families/species that experienced poleward shifts in their geographical distribution. For region 1, I looked for families present only in recent surveys (category 4) and exclusively represented by species with a tropical latitudinal range. For region 2 I looked for families of category 4 and also sampled in region 1. I wanted to consider families originally found in New South Wales that may have expanded their range to include Tasmanian waters.

(4) Families of category 1 to determine whether there were clear signs of extirpation.

4.3.6 Standardized indices of abundance for region 2

Variation in catch rates (or frequency of occurrence) between the two periods considered may be attributable to changes in sampling and/or environmental factors affecting availability to the survey gear as well as to changes in abundance (or presence) due to fishing or range shifts (Maunder & Punt, 2004). To account for variation in the data not attributable to changes in fish abundance, I standardized the catches by accounting for the sampling intensity and other environmental variables deemed to affect the variability of the catches. A standardized index of abundance for the *before* and *after* periods was calculated for gurnards (*Triglidae*), the most abundant family during the *before* period, and for flatheads (*Platycephalidae*) and morwongs (*Cheilodactylidae*), the main families targeted by a range of fisheries throughout the history of commercial fishing in South East Australia.

For each of these families I chose the most appropriate distribution describing the catch data, given as number of individuals in each tow, and used generalized linear models (GLMs) for the standardization process. The choice of GLMs was based on these regression models' ability to accommodate unbalanced sampling schemes (e.g. different number of surveys' tows carried out in each period) and non-normal data (i.e. species' catch distributions described by Negative Binomial, instead of Gaussian distributions).

Catch distributions for gurnards and flatheads were highly skewed and successfully described by a negative binomial distribution (NBD). Thus, for these families, I assumed that the probability (q) of obtaining y individuals of a family in tow_i was:

$$q(y|\mu,k) = \frac{\Gamma(k+y)}{\Gamma(a)\Gamma(y+1)}\left(\frac{k}{k+\mu}\right)^{k}\left(\frac{\mu}{k+\mu}\right)^{y} \qquad (4\text{-}1)$$

for $y = 0, 1, 2, \ldots$

where μ is the mean and k is the dispersion parameter, respectively. Both parameters are estimated from the data. In a negative binomial regression model a loglinear function relates covariates to μ of the NBD. I modeled $\log(\mu_i)$ as:

$$\log(\mu_i) = \alpha + \beta * X_i + \log(A_i) \qquad (4\text{-}2)$$

where α is the intercept, X_i are raw vectors containing covariate values for the i^{th} observation, and A_i is the swept area, treated as an offset. The matrix of covariates, X, included the tow period (*before* and *after*), the type of gear used (i.e. headrope length), the season of the year when the tow was carried out [i.e. spring (Oct-Dec), summer (Jan-Mar), autumn (Apr-Jun) or winter (July-Sept)], and the tow's latitude and depth. The effect of mesh size could have not been considered because a unique mesh size was used in each period (i.e. 76 mm during the *before* period and 90 mm during the *after* period), thus the effect of mesh size would have been confounded with that of tow period.

Catch distributions for morwongs were highly skewed and characterized by high presence of zero catches. For this reason I modeled these data with a zero inflated negative binomial distribution (ZINBD) (e.g. Minami *et al.,* 2007).

The ZINBD is a mixture of two distributions. The binomial distribution is used to model the probability (p) that a 'false zero' (e.g. no catch because of sampling errors in the course of data collection) is observed, and the NBD is used to model the probability (1-p) that either counts or 'true zeros' (i.e. no catch because of low species detectability) are observed (Martin *et al.,* 2005). The probability function for the ZINBD is expressed as:

$$(y_i|X_i, G_i, \beta, \gamma, k) = \begin{cases} p_i + (1-p_i)q(0|\mu_i, k), & y_i = 0 \\ (1-p_i)q(y_i|\mu_i, k) & , \ y_i = 1, 2, \ldots \end{cases} \qquad (4\text{-}3)$$

where $q(y_i| \mu_i, k)$ is given by (4-1).

In a ZINBD regression model a logistic regression function is used to relate covariates to p (probability of 'false zero'), and a loglinear function is used to relate covariates to μ of the NBD (modeling the probability of counts or 'true zero'). I modeled log(μ_i) as in (4-2), and logit(p_i) as:

$$\text{logit}(p_i) = \log\frac{p_i}{1-p_i} = \alpha + \gamma * G_i \qquad (4\text{-}4)$$

where α is the intercept and G_i are raw vectors containing covariate values for the i^{th} observation. Similar to X, the matrix of covariates G included tow's headrope length, season, latitude and depth. This is because the probability of 'false zero' may depend on sampling errors, such as net inefficiency in catching individuals of the morwongs, or sampling in seasons, depth and latitudes outside the temporal and spatial distribution of this family.

My main focus was to quantify the direct effect of single covariates (period, headrope, season, latitude and depth) on the catch rate of each family. However, to increase models' explanatory power and to better investigate the process driving family abundance, I also considered models including potentially important interactions between covariates (i.e. latitude*season and depth* season, which would allow me to detect seasonal changes in latitudinal and depth distribution).

For each family, I fitted a model including main effects and first order interactions of season with depth and latitude. This was for eqn. 4-2 as well as 4-4. Then, I explored the fit of models with different combinations of covariates and selected the model with the lowest Akaike Information Criterion (AIC, Akaike, 1974) as the best model. Lastly, from the best fitting model I predicted standardized indices of abundance for the *before* and *after* period. Standardized indices of abundance were predicted by setting all continuous covariates (i.e. tow's depth, latitude and longitude) at their means and all discrete covariates (i.e. headrope length and season) at their most common value in the dataset. Swept area, treated as an offset in the models, was set at 1 km^2.

4.4 Results

4.4.1 Region 1

4.4.1.1 Shifts in main families catch composition

Overall the most frequently caught families remained the same across the two periods (Table 4-1), with flatheads and gurnards being the families with the highest frequency of occurrence in both periods. Dories (*Zeidae*), skates (*Rajidae*), stingarees and whitings (*Sillaginidae*) were caught in similar frequencies between periods. Also, leatherjackets were frequently (>50% of tows) caught in both periods, but for this family the frequency increased by 45±1%.

A few families were representative of the *before* period's catch and not of the *after* period's catch (Fig. 4-3). These were sand and lefteye flounders (*Paralichthyidae* and *Bothidae*, respectively), whose frequency of occurrence dropped by 74±4% and 90±3%, respectively. Notably, these families showed the greatest change in frequency of occurrence among all families considered in this section. Neither family was of commercial value and both included species characterized by either high or medium resilience (Froese & Pauly, 2015). In contrast, guitarfishes (*Rhinobatidae*), alfonsinos

(*Berycidae*) and angelsharks (*Squatinidae*) were more representative of the *after* than the *before* period’s catch, and these families’ frequencies of occurrence increased by 37±4%, 41±4% and 49±3%, respectively. All these families were of commercial value and guitarfishes and angelsharks included species characterized by low resilience (two out of three and one out of two species sampled, respectively) (Froese & Pauly, 2015).

Table 4-1. Shifts in main families catch composition for region 1. Families are ordered according to frequencies of occurrence during the *before* period, from the most to the least frequent. Codes: B=before, A=after, Y=yes, N=no, L=low, M=medium, H=high, DE=demersal, BP=benthopelagic, RA=reef-associated, TR=tropical, S=sub-tropical, TE=temperate, TH=other.

Family	Species	Freq. before	Freq. after & SD	Main catch	Change & SD (%)	Commercial	Max size (cm)	Trophic level	Resilience	Column position	Lat. range
flatheads *(Platycephalidae)*		84	98;1	B & A	15;1	Y					
	Platycephalus richardsoni		92;2				65	3.9	M		
	Platycephalus bassensis		11;2				46	4.3	M	DE	TE
	Platycephalus conatus		9;2				70	4.2	L		
	Platycephalus caeruleopunctatus		7;2				45	3.4	M	DE	TE
	Thysanophrys cirronasa		5;2				38	3.9	H	DE	S
	Platycephalus endrachtensis		2;1				45		M		
	Platycephalus fuscus		2;1				120	4.1	M	DE	S
gurnards *(Triglidae)*		81	93;2	B & A	12;2	Y					
	Pterygotrigla polyommata	19	35;3				62	3.2	M	DE	TE
	Chelidonichthys kumu		50;3				60	3.7	H		
	Lepidotrigla vanessa		19;3				28	3.2	H	DE	TE
	Lepidotrigla modesta		13;2				22	3.5	H	DE	TE
	Lepidotrigla mulhalli		4;1				20	3.2	H	DE	TE
sand flounders *(Paralichthyidae)*		70	18;3	B	-74;4	N					
	Pseudorhombus jenynsii	58					34	3.5	H	DE	S
	Pseudorhombus arsius	51					45	4.2	M	DE	TR
	Pseudorhombus tenuirastrum	5					25	3.5	H	DE	TE
dories *(Zeidae)*		70	62;3	B & A	-11;5	Y					
	Zeus faber	70	61;4				90	4.5	M	BP	TE

Table 4-1. Continued

Family	Species	Freq. before	Freq. after & SD	Main catch	Change & SD (%)	Commercial	Max size (cm)	Trophic level	Resilience	Column position	Lat. range
skates *(Rajidae)*		65	61;3	B & A	-6;5	Y					
	Dipturus australis		11;2				50	3.5	M	DE	TH
	Dipturus confusus		6;2				65		M		
	Dipturus cerva		2;1				60	3.5	M	DE	TH
lefteye flounders *(Bothidae)*		56	5;2	B	-90;3	N					
	Lophonectes gallus	56	6;2				20	3.5	H	DE	TE
stingarees *(Urolophidae & Plesiobatidae)*		56	48;3	B & A	-13;6	N					
	Urolophus cruciatus	56					50	3.4	L	DE	S
	Urolophus bucculentus		30;3				80	3.6	L	DE	S
	Urolophus viridis		18;3				44	3.5	L	DE	S
	Trygonoptera testacea		9;2				47	3.8	L	DE	S
leatherjackets *(Monacanthidae)*		51	93;2	B & A	45;1	Y					
	Nelusetta ayraud	46	88;2				100	3.7	M	DE	S
	Eubalichthys mosaicus	2					60	2.8	L	DE	TE
	Meuschenia scaber	2					31	3	M	DE	TE
whitings *(Sillaginidae)*		51	61;3	B & A	16;5	Y					
	Sillago bassensis	51					33	3.3	M	DE	S
	Sillago flindersi		52;3				32	3.3	H	DE	S
guitarfishes *(Rhinobatidae)*		33	52;4	A	37;4	Y					
	Aptychotrema rostrata		37;3				100	3.9	L	DE	S
	Aptychotrema vincentiana		2;1				79	3.8	M	DE	S
	Trygonorrhina fasciata		2;1				126		L		
alfonsinos *(Berycidae)*		28	48;3	A	41;4	Y					
	Centroberyx affinis	28	48;3				51	3.8	M	BP	S
angelsharks *(Squatinidae)*		26	50;4	A	49;3	Y					
	Squatina australis		31;3				152	4	L	DE	S
	Squatina albipunctata		7;2				98		M		

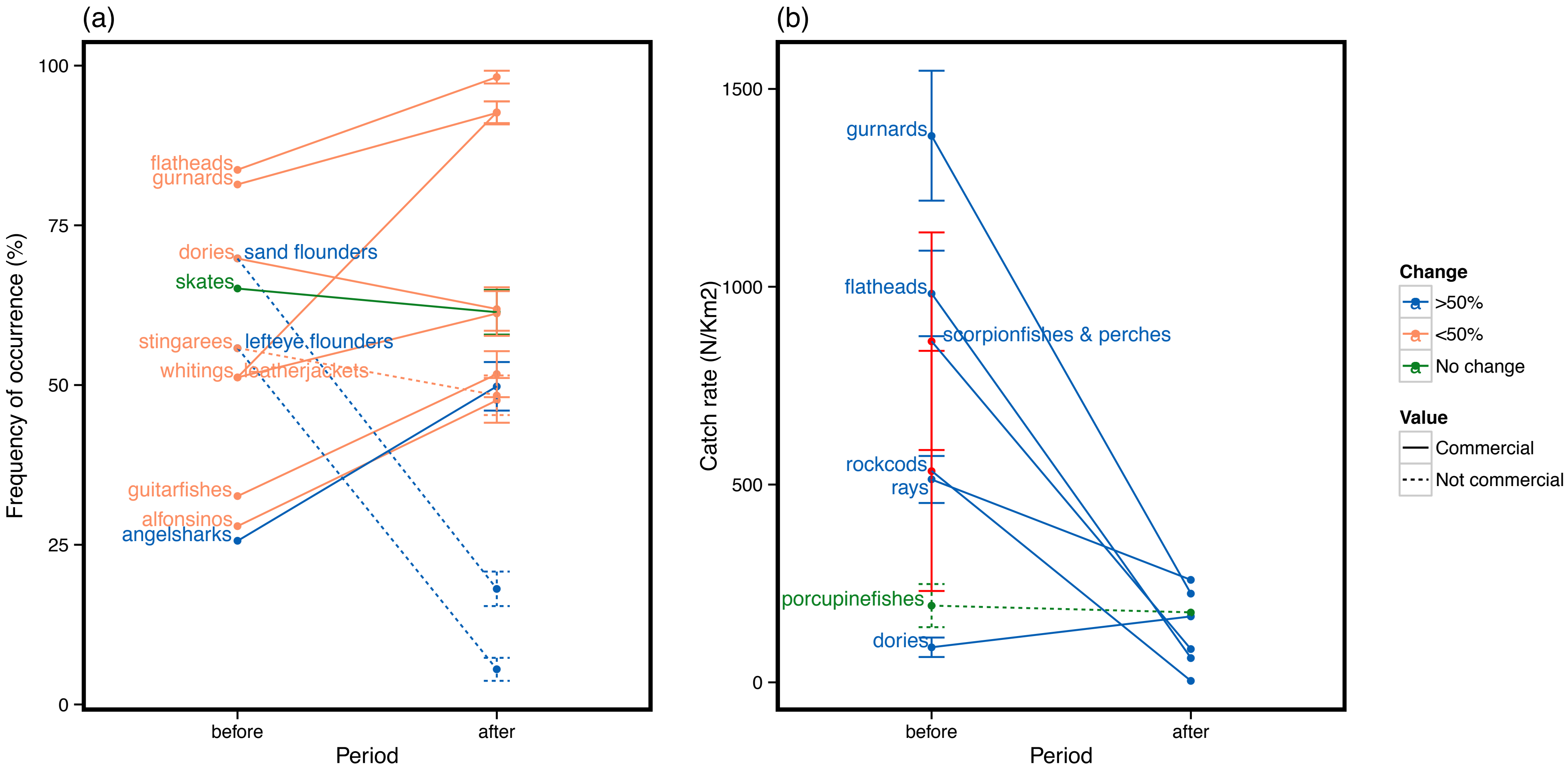

Figure 4-3. Shift in catch composition, and changes in (a) frequency of occurrence for region 1 and (b) catch rates for region 2 for the main families sampled. In (b) high standard deviation are shown in red for the families “scorpionfishes & perches” and rockcods.

4.4.1.2 Marked changes in less frequently caught families

Among the families of categories 2 and 3 and sampled with a frequency of occurrence between 20% and 50%, decreases over a 50% threshold were reported for righteye flounders (*Pleuronectidae*) (93±3%), lizardfishes & relatives (69±6%), grubfishes (*Pinguipedidae*) (61±9%), bellowfishes (*Macroramphosidae*) (60±9%), boarfishes (*Pentacerotidae*) (58±8%), boxfishes (*Ostraciidae*) (57±5%) and eagle rays (*Myliobatidae*) (54±8%) (Table 4-2). None of these families were of commercial value, although flounders and grubfishes can be caught in large number as by-product of trawling or by recreational anglers (Rowling *et al.,* 2010), and they all included species characterized by either high or medium resilience. On the other hand, frequencies of occurrence of conger eels, porcoupinefishes and morwongs increased by 89±1%, 87±1% and 64±3%, respectively. Of these families, morwongs are a primary target family of a range of fisheries and a frequently caught species belonging to this family, *Nemadactylus douglasii*, is characterized by a low resilience.

4.4.1.3 Poleward shifts in the geographical distribution of species

Some species primarily associated with tropical environments were sampled during each period. However, none of the families of category 4 exclusively included species with a tropical latitudinal range.

All families of categories 1 and 4 were characterized by very low frequencies of occurrence (below 10%), thus rarely sampled even in the one period in which they were found. The only exception was the stargazer (*Uranoscopidae*) family, of category 1 and with a frequency of occurrence of 23.3% in the *before* period.

Table 4-2. Marked changes in less frequently caught families for region 1. Families are ordered according to the magnitude of change in frequencies of occurrence between the *before* and *after* period, from steep decreases to steep increases. Codes as per Table 4-1.

Family	Species	Freq. before	Freq. after & SD	Change & SD (%)	Commercial	Max size (cm)	Trophic level	Resilience	Column position	Lat. range
righteye flounders *(Pleuronectidae)*		28	2;1	-93;3	N					
	Ammotretis rostratus	2				30	3.1	M	DE	TE
lizardfishes & rel. *(Chlorophthalmidae & Paraulopidae & Bathysauroididae & Bathysauropsidae & Synodontidae)*		30	9;2	-69;6	N					
	Trachinocephalus myops	2	4;1			23	4.4	M	RA	TE
	Saurida undosquamis		4;1			50	4.5	H	RA	S
grubfishes *(Pinguipedidae)*		23	9;2	-61;9	N					
	Parapercis allporti		9;2			33	3.4	H	DE	TE
bellowsfishes *(Macroramphosidae)*		23	9;2	-60;9	N					
	Macroramphosus scolopax	23	4;1			20	3.5	H	DE	TE
boarfishes *(Pentacerotidae)*		30	13;2	-58;8	N					
	Zanclistius elevatus	28	7;2			33	3.4	H	DE	TE
	Paristiopterus labiosus	2	5;1			100	3.3	M	DE	TE
boxfishes *(Ostraciidae)*		39	17;2	-57;5	N					
	Anoplocapros inermis		7;2			37	3.1	TH	DE	S
eagle rays *(Myliobatidae)*		23	15;2	-54;8	N					
	Myliobatis australis		13;2			120	3.6	M	DE	S
morwongs *(Cheilodactylidae)*		14	39;3	64;3	Y					
	Nemadactylus douglasii	14	33;3			81	3.5	L	DE	TE
	Nemadactylus macropterus		9;2			70	3.4	M	DE	S
porcupinefishes *(Diodontidae)*		5	36;3	87;1	N					
	Dicotylichthys punctulatus		11;2			40	3.5	TH	RA	S
	Allomycterus pilatus		11;2			50	3.4	TH	DE	TE
conger eels *(Congridae & Colocongridae)*		2	20;2	89;1	N					

4.4.2 Region 2

4.4.2.1 Shifts in main families catch composition

The total catch rate across all species and families decreased fourfold between the *before* and *after* periods (from 7015±612 N/km^2 in 1909-1910 to 1608 N/ km^2 in 1979-1983). Notably, in the historical surveys 47% of the tows sampled more than 300 individuals of at least one family (e.g. a single tow reported 1818 individuals of the gurnard family), whereas none of the tows carried out during the *after* period reported such high catches per family. Because families' catch rates were calculated multiple times using a random selection of tows (see section "Indices of presence and abundance"), tows reporting extraordinary catches during the *before* period were not considered for each calculation. This resulted in particularly high standard deviations of catch rates for some families (Table 4-3 and Fig. 4-3(b)).

The composition of the catch markedly changed between the two periods considered (Table 4-3 and Fig. 4-3). In the *before* period gurnards, flatheads, "scorpionfishes & perches" and rockcods (*Serranidae*) accounted for about 50% of the total catch whereas in the *after* period rays were the most abundant group of families, and, together with gurnards, porcupinefishes and dories accounted for about 50% of the catch. Although gurnards were among the most abundant group of families in both periods, their catch rate steeply declined from 1371±174 N/ km^2 in the *before* period to 224 N/ km^2 in the *after* period. Steep declines in catch rate between the two periods were also reported for other families, including the most abundant of the *after* period (i.e. rays). In particular, rockcods, flatheads, "scorpionfishes & perches" and rays decreased by 99±0.5%, 94±0.7%, 90±2.3% and 50±5.3%, respectively. By contrast, only the dories family has shown an increase between the two periods (47±15.7%).

All families considered in this section, except porcupinefishes, were of commercial values. The family "scorpionfishes & perches" included one species (*Helicolenus percoides*), over the six sampled, characterized by low resilience, and the family rays included four species, over the 12 sampled, characterized by low resilience (Froese & Pauly, 2015).

Table 4-3. Shifts in main families catch composition for region 2. Families are ordered according to catch rates during the *before* period, from the highest to the lowest. Codes as per Table 4-1.

Family	Species	Catch rate before & SD	Catch rate after	Main catch	Change & SD (%)	Commercial	Max size (cm)	Trophic level	Resilience	Column position	Lat. range
gurnards *(Triglidae)*		1382;164	225	B & A	-84:2	Y					
	Lepidotrigla spp.	1354;151									
	Lepidotrigla modesta		132				22	3.5	H	DE	TE
	Lepidotrigla vanessa		60				28	3.2	H	DE	TE
	Pterygotrigla polyommata		20				62	3.2	M	DE	TE
	Lepidotrigla mulhalli		11				20	3.2	H	DE	TE
	Lepidotrigla papilio		1				20	3.5	H	DE	TE
	Chelidonichthys kumu		1				60	3.7	H		
flatheads *(Platycephalidae)*		983;108	62	B	-94	Y					
	Platycephalus richardsoni		53				65	3.9	M		
	Platycephalus bassensis		8				46	4.3	M	DE	TE
	Rogadius patriciae		1				27	3.7	H		
scorpionfishes & perches *(Scorpaenidae & Sebastidae & Neosebastidae)*		862;275	84	B	-90;2	Y					
	Neosebastes spp.	504;225									
	Helicolenus percoides	125;37					47	4	L	DE	S
	Neosebastes pandus		45				50	3.7		DE	S
	Neosebastes thetidis		24				35	3.6		DE	TE
	Scorpaena papillosa		15				30	4	M	DE	TE
	Neosebastes occidentalis		0.1				18				

Table 4-3. Continued

Family	Species	Catch rate before & SD	Catch rate after	Main catch	Change & SD (%)	Commercial	Max size (cm)	Trophic level	Resilience	Column position	Lat. range
rockcods *(Serranidae)*		534;303	4	B	-99	Y					
	Caesioperca rasor	512;300	0.4				25	3.5	H	DE	TE
	Lepidoperca pulchella	6;4					28	3.1	H	DE	TE
	Caesioperca lepidoptera	1;1	3				30	3.1	H	DE	TE
	Lepidoperca spp.		0.1				34	3.6			
rays		513;59	259	A	-49;5	Y					
	Urolophus cruciatus		77				50	3.4	L	DE	S
	Urolophus paucimaculatus		55				57	3.7	L	DE	S
	Dipturus cerva		38				60	3.5	M	DE	TH
	Urolophus bucculentus		19				80	3.6	L	DE	S
	Zearaja spp.		19								
	Urolophus viridis		17				44	3.5	L	DE	S
	Pavoraja nitida		13				37	4	M	DE	S
	Dentiraja lemprieri		10				52	4	M		
	Narcine tasmaniensis		9				47	3.3		DE	TH
	Spiniraja whitleyi		1								
	Myliobatis australis		0.2				120	3.6	M	DE	S
	Dasyatis brevicaudata		0.1				430	3.9		DE	TE
porcupinefishes *(Diodontidae)*		194;54	177	A	-9;20	N					
	Allomycterus pilatus		106				50	3.4		DE	TE
	Diodon nicthemerus		71				40	3.5		DE	TE
dories *(Cyttidae & Zeidae)*		89;25	167	A	47;14	Y					
	Cyttus australis	87;25	95				41	3.5		DE	TE
	Zeus faber	2;1	8				90	4.5	M	PB	TE
	Cyttus novaezealandiae		64				40	3.3		DE	TE

4.4.2.2 Marked changes in less frequently caught families

Among the families considered in this section splendid perches (*Callanthiidae*) and whitings showed the steepest decline in catch rate. Both families were virtually absent in the recent surveys. Steep decreases in catch rates were also reported for “flounders & soles” (91±2.3%), leatherjackets (88±2.3%), and morwongs (87±2%) (Table 4-4). Of these families, whitings, leatherjackets and morwongs were of commercial value. Leatherjackets and morwongs included species characterized by low resilience (one out of the six and one out of the three species sampled, respectively). On the other hand, only the boxfish family has shown an increase over the 50% threshold between the two periods. This was by 85±4%.

When considered together, strictly commercial families (i.e. families such as flatheads and morwongs for which the majority of species sampled were of commercial value) decreased from 2507±249 N/km^2 during the *before* period to 555 N/km^2 during the *after* period. Also mixed families (i.e. families such as sharks and rays including species of and not of commercial value) showed a similar decrease. On the other hand, strictly not commercial families (i.e. families such as porcupinefishes for which none of the species sampled were of commercial value) showed a more stable catch rate in time, from 482±255 N/km^2 during the *before* period to 314 N/km^2 during the *after* period. Summarizing, I found steep declines in groups of commercial families, and a much less decline in the group of non-commercial families.

Table 4-4. Marked changes in less abundant families for region 2. Codes are as per Table 4-1.

Family	Species	Catch rate before & SD	Catch rate after	Change & SD (%)	Commercial	Max size (cm)	Trophic level	Resilience	Column position	Lat. range
splendid perches *(Callanthiidae)*		400;60	1	-100	N					
	Callanthias spp.	408;67								
	Callanthias allporti		1			30	3.4	TH	RA	TE
	Sillago spp.	419;98				23	3.2			
whitings *(Sillaginidae)*		408;88	4	-99	Y					
	Sillago bassensis	19;9	4			33	3.3	M	DE	S
flounders & soles *(Pleuronectidae & Soleidae & Bothidae & Paralichthyidae)*		110;32	10	-91;2	N					
	Ammotretis rostratus	6;4	7			30	3.1	M	DE	TE
	Rhombosolea tapirina	4;2	2			45	3	M	DE	S
	Pseudorhombus jenynsii	0.5;0.4				34	3.5	H	DE	S
	Pseudorhombus arsius	0.2;0.2				45	4.2	M	DE	TR
	Ammotretis lituratus		0.1			23	3.6	H	DE	TE
leatherjackets *(Monacanthidae)*		351;77	41	-88;2	Y					
	Nelusetta ayraud	341;80				100	3.7	M	DE	S
	Meuschenia scaber		21			31	3	M	DE	TE
	Anacanthus barbatus		9			35	3	M	RA	TR
	Eubalichthys mosaicus		7			60	2.8	L	DE	TE
	Acanthaluteres vittiger		4			35	2	M	DE	TE
	Meuschenia australis		0.1			30	3	M	DE	TE
morwongs *(Cheilodactylidae)*		519;78	68	-87;2	Y					
	Nemadactylus macropterus	532;87	68			70	3.4	M	DE	S
	Nemadactylus douglasii	0.3;0.3				81	3.5	L	DE	TE
	Cheilodactylus nigripes		0.1			41	3.1	M	RA	TE
boxfishes *(Ostraciidae)*		12;3	78	85;3	N					
	Aracana aurita		78			20	3.1	TH	DE	TE
	Aracana ornata		0.6			15	3	TH	DE	TE

4.4.2.3 Poleward shifts in the geographical distribution of species

Families of region 2 and category 4 also sampled in region 1 were toadfishes (*Tetraodontidae*), pipefishes (*Syngnathidae*) and grubfishes (*Pinguipedidae*) (Table 4-5). In particular, two species of toadfishes, *Contusus richie* and *Omegophora armilla*, one species of pipefishes, *Solegnathus robustus*, and one individual of grubfishes, *Parapercis allporti,* were sampled in region 2. Among all species, only *Omegophora armilla,* is known to have recently (between the 1980s and present) expanded its range and abundance in southeast Tasmania (Last *et al.,* 2011). On the contrary, *Parapercis allporti* (one individual sampled) is endemic to South East Australia (Atlas of Living Australia, 2015), thus a shift in latitudinal range was excluded.

Similar to region 1, all families of categories 1 and 4 were characterized by very low catch rates (less than 10 N/km^2). Exceptions were the families southern hakes (*Macruronidae*) and toadfishes, of category 4 and with a catch rate of 25 and 18 N/km^2, respectively.

Table 4-5. Poleward shift in the geographical distribution of species for region 2. Codes as per Table 4-1.

Family	Species	Freq. before	Freq. after & SD	Catch rate after	Max size (cm)	Trophic level	Column position	Lat. range
toadfishes *(Tetraodontidae)*		7	11;2	18				
	Contusus richei			11	25	3	DE	TE
	Omegophora armilla			7	25	3.3	DE	TE
	Lagocephalus lunaris	5			45	3.4	DE	TR
	Lagocephalus sceleratus		2;1		110	3.6	RA	TR
	Solegnathus robustus			3	35	3.5	DE	TH
pipefishes *(Syngnathidae)*		9	7;2	3				
	Phyllopteryx taeniolatus		2;1		46	3.5	DE	TE
grubfishes *(Pinguipedidae)*		23	9;2	0.1				
	Parapercis allporti		9;2	0.1	33	3.4	DE	TE

4.4.2.4 Comparison between frequency of occurrence and catch rates

For region 2, I looked at the most frequently encountered (frequency of occurrence >50%) and the most abundant (accounting cumulatively for 50% of the total catch) families (Fig. 4-4 (a) and (b), respectively), and I found divergences between these two indices. In general, whereas catch rates showed steep declines for almost all families considered (as reported in section "Shifts in main families catch composition"), frequencies of occurrence gave a less clear picture. Importantly, frequencies of occurrence underestimated both declines for some families (i.e. gurnards, leatherjackets, flatheads, morwongs, "scorpionfishes & perches", and rays) and increases for others (i.e. dories and stargazers). For same families the direction of changes differed between the two indices considered, i.e. sharks and porcupinefishes decreased if I was to consider catch rates and increased if I was to considered frequency of occurrence, although in all cases the change was marginal. Also, the main families catch composition differed depending on which of the two indices I considered. For example, for the *before* period frequencies of occurrence showed higher dominance of rays, morwongs and sharks, and lower dominance of rockcods, than catch rates.

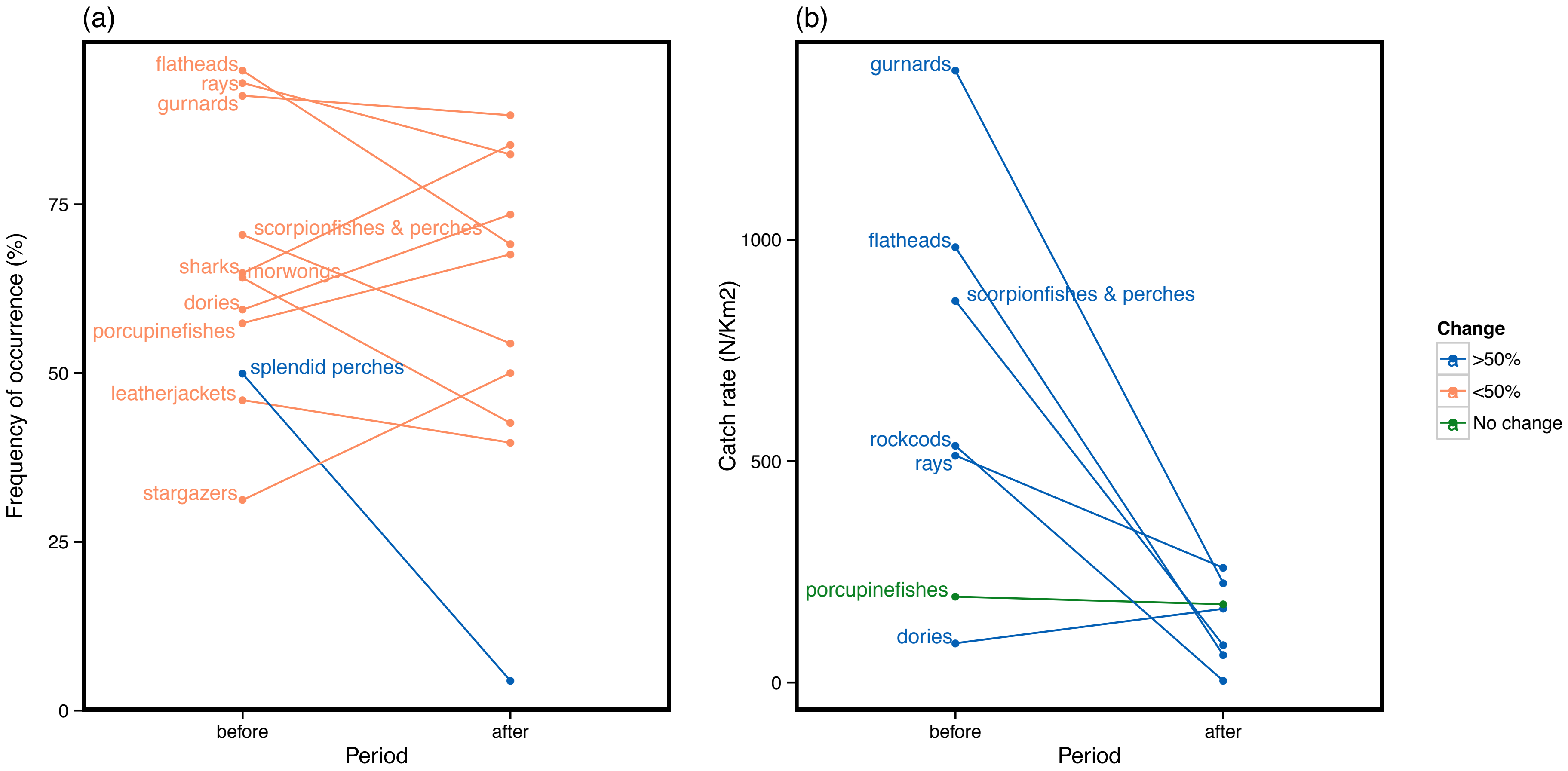

Figure 4-4. Shift in catch composition and changes in (a) frequency of occurrence and (b) catch rates for the main families sampled in region 2, between the *before* and *after* periods. For clarity, standard deviations are not shown, but see Fig. 4-3 (b).

4.4.2.5 Standardized indices of abundance

The GLM models fitted to counts data for gurnards, flatheads and morwongs showed that these family's abundance significantly decreased between periods (Table 4-6). Other covariates influenced these families' catches, although generally to a much lesser extent. In particular, tow depth had a positive effect on gurnards' abundance, indicating that higher catches of this family were recorded in deeper waters of the continental shelf. In contrast, headrope length had a negative effect on both gurnards' and flatheads' abundance, suggesting that nets with wider mouths may less efficiently catch individuals of these families. Also, season strongly influenced flatheads' and morwongs' abundance, with lowest catches of flatheads recorded in summer and lowest catches of morwongs recorded in winter. For morwongs, depth and headrope length were marginally significant ($p<0.05$) and their (positive) effects seemed weaker than for the covariates of period and season. Also, there was not a significant effect of latitude, although excluding this covariate decreased the model fit (this was for both eqn. 4-2 and 4-4). For this family (modeled using a ZINBD) the probability of 'false zero' decreased with (increasing) depth and was higher for larger nets, thus indicating that tows carried out in shallow waters of the continental shelf or with nets with wide mouths were likely to report null catches of the morwong family, but these null catches did not suggest absence of morwongs.

There was no strong evidence that the inclusion of interactions (season*depth, season*lat) improved the models for any of the families considered.

Table 4-6. GLM results for the families gurnards, flatheads and morwongs.

Gurnards				
Parameter	Estimate	SE	t	*P*-value
intercept	6.958	0.111	62.484	<0.001
after	-1.474	0.211	-6.984	<0.001
depth	0.409	0.093	4.393	<0.001
headrope	-0.4	0.095	-4.199	<0.001

Flatheads				
Parameter	Estimate	SE	t	*P*-value
intercept	7.113	0.231	30.779	<0.001
after	-2.432	0.244	-9.984	<0.001
spring	-0.208	0.264	-0.789	0.43
summer	-1.151	0.431	-2.667	<0.01
fall	-0.359	0.275	-1.305	0.1918
headrope	-0.251	0.102	-2.471	<0.05

Morwongs - counts				
Parameter	Estimate	SE	z-value	*P*-value
intercept	-4.245	6.343	-0.669	0.8598
after	-2.228	0.532	-4.19	<0.001
spring	2.056	0.52	3.95	<0.001
summer	2.276	0.649	3.508	<0.001
fall	2.281	0.469	4.868	<0.001
depth	0.014	0.005	2.559	<0.05
latitude	-0.103	0.135	-0.759	0.4478
headrope	0.117	0.048	2.413	<0.05
log(theta)	-1.125	0.129	-8.718	<0.001
Morwongs - zeros				
Parameter	Estimate	SE	z-value	*P*-value
intercept	22.223	13.69	1.623	0.1045
depth	-0.106	0.03	-3.553	<0.001
latitude	0.689	0.396	1.743	0.0814
headrope	0.316	0.156	2.028	<0.05

Model diagnostics for the gurnard and flathead families (See Figs 9-1 and 9-2 in Appendix 2) indicated that the conditions of homogeneity, independence and normality were overall met. An apparent pattern in the residual was a striation due to tows reporting null catches.

The plot of Pearson residuals versus fitted values for the morwong family showed expected patterns for highly overdispersed counts data (see Fig. 9-3 in Appendix 2). The plot of observed versus predicted counts highlighted the model's inability to predict exceptional catches during the *before* period.

Standardized indices of abundance (Fig. 4-5) assumed longitude of 147.8 DD East, latitude of 41 DD South, depth of 69 m and using a net with headrope length of 29 m. Season was set as 'fall' for the flathead and morwong families. Notably, confidence intervals for standardized indices of abundance were high compared to the mean, indicating that these values were not precisely estimated. Despite uncertainties, results showed a general agreement between the un-standardized catch rate and the standardized index of abundance for all families considered in this section (Fig. 4-5).

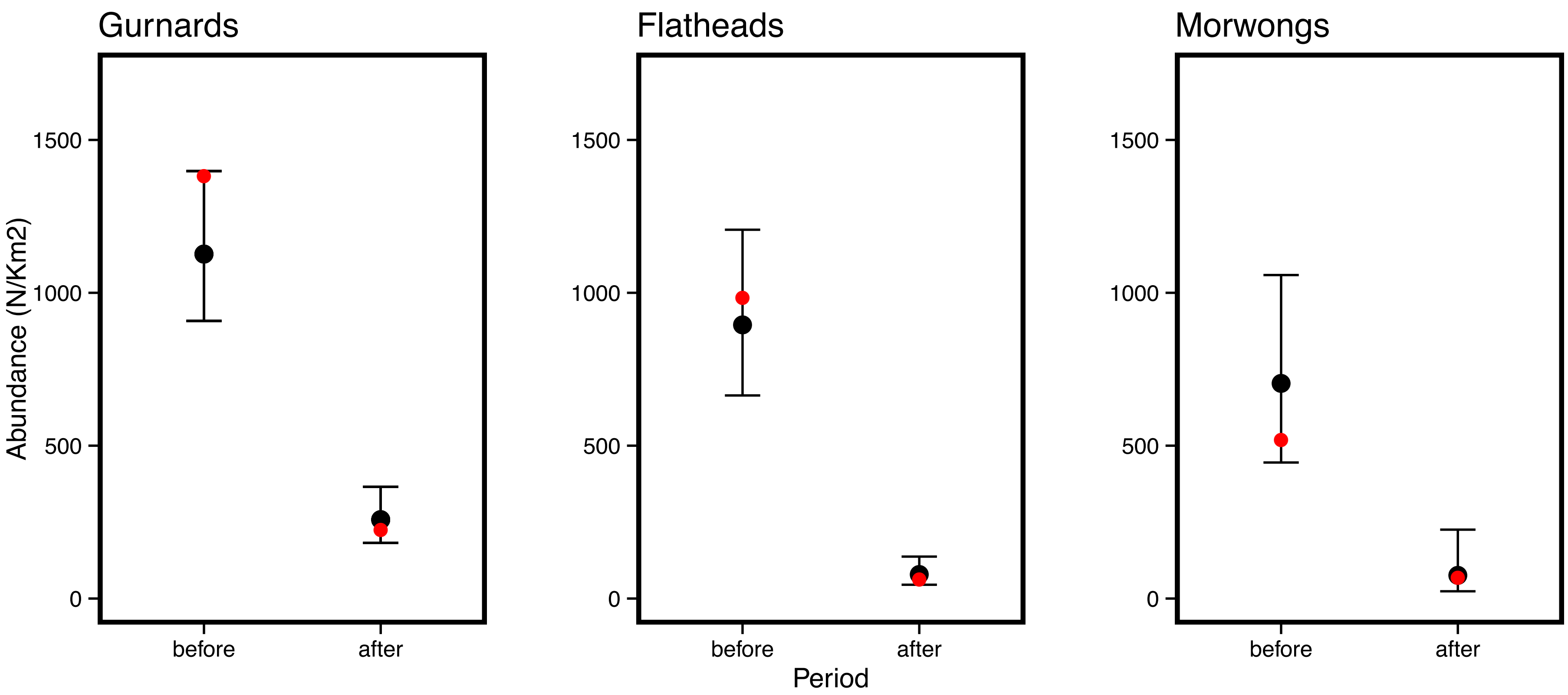

Figure 4-5. Expected counts for gurnards, flathead and morwong families at mean values of latitude, longitude and depth across the dataset and at values of headrope length of 29 m. For the flathead and morwong families season was set as ‘fall’. Confidence intervals at 95% are also shown. Red dots indicate nominal catch rates.

4.5 Discussion

This study found strong evidence for changes in the structure of demersal fish communities of South East Australia over the span of about a century. In particular, marked changes were found for demersal fish communities of the continental shelf around Tasmania, between 1909-1910 (*before* period) and 1978-1983 (*after* period). These included shifts in the catch composition of the main families as well as sharp declines in the total and individual family catch rates. Also, standardized indices of abundance for gurnards, flatheads and morwongs confirmed the declining trend after the influence of a variety of sampling and environmental factors (e.g. shifts in sampling depth and latitude) was removed from observed catches.

Notably, steep decreases in Tasmanian demersal fish catch rates and standardized abundances have been detected for all main commercial families (except dories, Table 4-3, and Fig. 4-3). Fishing is expected to reduce target and by-catch species abundance, through the consistent removal of individuals and potentially also through habitat modification (e.g. Dayton *et al.,* 1995; Jennings & Kaiser, 1998; Watling & Norse, 1998). Hence, it is most likely that some of the decreases were due to the impact of fishing on fish stocks, and, more broadly, on demersal communities.

The combined effects of local fisheries using a number of non-trawl methods (e.g. hook and lines, seine and gill-nets) as well as those of trawling, which was mostly concentrated further north but could have nonetheless impacted Tasmanian resources if target stocks spanned a wider region, may have been responsible for the steep decline in gurnards, flatheads and morwongs, targeted since the 1880s (Johnston, 1882; Klaer, 2004; Bridge, 2009; André *et al.,* 2015). These patterns are in accordance with the findings of related studies, which have shown marked declines in total catch rates and in the abundance of the main commercial and by-catch species on the continental shelf and slope of New South Wales, when data were analyzed spanning the first phase of trawling (1915-1961) or collected before and after its expansion to deeper waters (1976-1996) (Andrew *et al.,* 1997; Klaer, 2001). Further, stock assessments for the main species of flatheads (tiger flathead, *Neoplatycephalus richardsoni*) and morwongs (jackass morwong, *Nemadactylus macropterus*) confirmed that these stocks declined between the early 1900s and the 1980s in the South East Australian region (Klaer, 2010; Wayte & Fay, 2011).

Fishing will have also altered community size structure, through the removal of larger individual fishes across a range of species (Pauly *et al.,* 1998; Pinsky & Byler, 2015). This was suggested by the substantial decrease, on the continental shelf of Tasmania, in the abundance of jackass morwong, a species that can reach 70 cm in length and 50 years of age (Table 4-4) (Froese & Pauly, 2015). A steep decrease in the abundance of this and other large, long-lived species (such as sharks) was also reported on the continental slope of New South Wales, after the first 20 years of trawling (1976-1996) (Andrew et al., 1997). Likewise, the analysis of a series of historical records (e.g. regional fish guides) showed range reduction and regional extirpation of some large teleosts, such as the blue morwong (*Nemadactylus valenciennesi*), in Tasmanian seas since the 1880s (Last *et al.,* 2011). All these studies confirmed that community size structure had changed and proposed that fishing was the likely cause.

Whereas the effect of fishing on demersal fish communities of Tasmania was evident, that of climate change, manifested as shifts in the latitudinal distributions of species (Johnson *et al.,* 2011; Last *et al.,* 2011), was minimal. My results showed that the effect of climate change only related to the increase in abundance of the ringed toadfish (*Omegophora armilla*, Table 4-5), which is known to have recently (between the 1980s and present) expanded its range and abundance in southeast Tasmania (Last *et al.,* 2011). This was despite increases in temperature in Tasmanian waters since the 1940s (Ridgway, 2007), so that climate-related changes in demersal fish communities might have been expected. The lack of stronger evidence may be due to the low taxonomic resolution of the data (families), but also to the stronger effect of fishing in shaping demersal fish communities, at least between the years considered in this study (1909-1910 to 1978-1983, for the Tasmanian shelf), as also suggested by Last *et al.,* (2011).

Although clear patterns emerged from my analyses and were consistent with those of other studies, limitations in the ability to compare historical and recent data may have biased some of the results. Some of these constraints were related to shifts in sampling depth and latitude between periods that were not accounted for when comparing nominal catch rates. For example, shift in sampling effort towards deeper waters of the continental shelf of Tasmania during the *after* period, may explain the increase in dories, which was driven by high abundance of the New Zealand dory (*Cyttus novaezealandiae*) that is commonly found on the outer shelf (Francis *et al.,* 2002). However, as I standardized the sampling between periods by defining overlapping areas (according to depth and latitude) and randomly selecting an equal number of tows for each period, shifts in sampling depths and latitudes may have been marginal and I conclude that these were not a major cause of change (except for particular cases, such as the dories and few other families whose distribution is chiefly determined by depth, e.g. flatheads).

Other constraints limiting the comparison of demersal fish communities over a large time scale were related to changes in catch efficiency of gear and survey vessels, selectivity of fishing nets, and environmental parameters, as well as to the natural variability of fish populations. All these factors are expected to influence catch rates (and standardized indices of abundance, Reeves *et al.,* 1992; Maunder & Punt, 2004; Willis & Birks, 2006), but I was unable to quantify or consider their possible effects. For example, whereas gear and survey vessel technology greatly improved between the *before* and *after* period (e.g. improved net material, adoption of GPS and ecosounder, Lyle, 1993), thus potentially leading to higher catch rates in later years, larger mesh size used during the *after* period (90 mm compared to 76 mm of the *before* period) could have emphasized the declines. This is because net mesh size affects the proportion of fish retained, with smaller mesh sizes retaining a larger proportion of (smaller) fish (Reeves *et al.,* 1992). However, it is unlikely that difference in these factors between periods can by themselves explain the strong decrease in catch rates and standardized indices of abundance, shared across most families and confirmed by multiple lines of evidence.

While I found strong shifts in community composition and sharp declines in most family abundances on the continental shelf of Tasmania, observed changes for the New South Wales region between 1898 and 2007-2011 were less apparent. Here I considered families' frequency of occurrence and found that this index was either stable or increased for all main commercial families (e.g. flathead, gurnards, leatherjackets, morwongs and whitings), despite some of these families including species with low resilience to fishing effort (e.g. leatherjackets and morwongs) and despite that the continental shelf of New South Wales had been extensively fished over the time considered (e.g. Tilzey & Rowling, 2001; Woodhams *et al.,* 2011).

These findings contrast with those of a similar study and of current stock assessments, as previously discussed (Klaer, 2001, 2004, 2010; Wayte & Fay, 2011). On the contrary, most of the non-target families' frequency of occurrence declined between periods.

On one hand, the unexpected pattern may be due to the different nature of the two datasets (*Thetis* and ISMP). For instance, the ISMP data (*after* period) may be biased by the commercial nature of the tows sampled, and by the tendency of fishers to trawl in areas where target species can be found. In such cases, declines in target families' frequency of occurrence may be masked because a family could have been frequently sampled despite possible decreases in abundance. Also, declines in non-target families' frequency of occurrence may be enhanced if, for example, target families are associated with habitats that do not support non-target families. In addition, improved fishing technologies may have enabled the trawl net to retained more of the target families and less of the non-target families during the *after* period. For instance, the *Thetis's* (*before* period) net was unable to retain some of the catch so that a consistent number of fish, including target families, was lost during each operation (Farnell & Waite, 1898).

On the other hand, an index of presence may be inadequate to detect temporal changes in demersal fish communities because a family could be detected with the same frequency over time, although its abundance may have changed. Using data from the continental shelf of Tasmania, I compared results from family catch rates and frequency of occurrence, and found differences between the two indices. For instance, frequencies of occurrence underestimated declines for some dominant families, such as gurnards, whereas for other families the directions of change differed according to the index considered. However, in general, if a family's catch rate steeply increased/decreased between the two periods, this change was also reflected in the family's frequency of occurrence, but to a lesser extent. This suggests that an index of presence can capture important changes in communities, thus making analysis of trends in community composition that mostly rely on species lists worthwhile.

Some of this study results may have direct application to assessment of fish stocks. For example, the assessment for tiger flathead considers individuals caught off Tasmania as a separate stock, because catches from this region contained a higher proportion of larger and older fish than those from the main northern fishing grounds off New South Wales, suggesting lighter exploitation or different population characteristics (Klaer, 2010). Whereas catch data for Tasmania are available for the period 1985-present and show stable catches through time, my results indicate that steep declines in the region preceded the availability of these data. The information provided in this study might be used to set an initial level of depletion at the start of the time series data (for the Tasmanian region) used in the assessment. This might improve current assessment for flatheads.

More importantly, the longer term historical data analyzed in this study improves our understanding of how the demersal fish communities of South East Australia were once structured, providing a baseline against which the magnitude and direction of more recent impacts can be measured. The comparison of historical bottom trawl surveys with their modern counterparts was not straightforward and required methods of analysis to deal with differences in the surveys over the period of a century or more, but the effort is worthwhile where it can give a clear picture of ecological changes at the whole community level and provide insight into the most likely cause of these changes, in this case due to fishing. This knowledge helps put the present status of demersal fish communities into better context, and may inform the setting of achievable and sustainable management goals for the future.

5 Chapter 5 – Species accumulation curves as indicators of human impacts on demersal fish communities

5.1 Abstract

The relationship between number of species and area is a well-known macro-ecological property that can be used as an indicator of human impact on natural communities. This relationship can be described by a species accumulation curve. I explored the application of species accumulation curves to datasets of bottom trawl surveys to determine their ability to capture fishing induced changes in demersal fish communities. First, I performed numerical simulations to investigate whether species accumulation curve was sensitive to changes in community properties, such as richness, species abundances and spatial distribution. Then, I analyzed a dataset of bottom trawl surveys carried out in South East Australia spanning 20 years. These data provided information on demersal fish communities at different stages of fishing exploitation. I built species accumulation curves along spatial and temporal gradients of community exploitation, and used the rate of species accumulation to characterize community structure. Linear mixed effects (LME) models were used to quantify the effect of fishing exploitation on species accumulation rates, while controlling for environmental and sampling factors. Numerical simulations showed that species accumulation curves can capture changes in community properties. Observed species accumulation rate decreased with increasing exploitation, and was sensitive to environmental variation along a latitudinal and depth gradient and to changes in sampling. These results indicate that species accumulation curves applied to trawl survey data can capture fishing induced changes in the structure of demersal fish communities. Therefore, I propose that species accumulation curves can be used to

inform on demersal fish communities' status and aid community monitoring. Furthermore, if applied to datasets of bottom trawl surveys available worldwide, species accumulation curves could be used to rank demersal fish communities for their level of exploitation.

5.2 Introduction

Species-area relationship (SAR), the relationship between number of species (*S*) sampled in a given area and the area sampled (*A*) (e.g. Arrhenius, 1921), is one of the strongest empirical laws in ecology (Schoener, 1976). This relationship has been widely used to estimate and predict a number of ecological processes from the theoretical number of species present in an area (Palmer, 1990; Gray, 2000), to evaluating species extinction rates from habitat loss (Pereira & Daily, 2006; Pereira *et al.,* 2010) and climate change (Thomas *et al.,* 2004), or to indicate human impact on natural communities (McClanahan, 1994; Flather, 1996; Tittensor *et al.,* 2007; to cite some examples).

SAR represent the increase in species richness of a community as the area investigated increases, reflecting three fundamental processes: as larger areas are sampled, there is a greater chance of detecting additional species (Arrhenius, 1921); larger areas encompass a greater and more diverse set of habitats, and thus include a greater and more diverse set of associated species (Williams, 1943); larger areas have higher rates of colonization and lower rates of extinction and thus promote higher diversity (MacArthur & Wilson, 1967; Connor & McCoy, 1979). However, SAR depends on sampling methods and on the techniques used to combine the data (Scheiner, 2003). It is influenced by the spatial arrangement of sampling units (i.e. nested, adjacent or spatially separated); and the technique used to combine them (e.g.

samples may be combined randomly or following the spatial order in which they were collected). The interplay of these factors may determine which aspects of diversity SAR is able to detect (Scheiner, 2003), and what function describe this relationship (Tjørve, 2003).Consequently, a variety of SARs is found in the literature (e.g. Scheiner, 2003; Dengler, 2009). Here I consider SARs generated from the accumulation of samples of the same size drawn from an expanse of habitat with boundaries defined by the survey design. These SARs are also called species accumulation curves (SACs).

Mathematically, among the structural equations that can define SAC (Flather, 1996; Tjørve, 2003), the most frequently used are power functions,

$$S = c * A^z \tag{5-1}$$

which can be linearized as log(S) = log(c) +z*log(A), and exponential functions,

$$S = c + z * \log(A) \tag{5-2}$$

where *S* is the number of species; *A* is the area; and *c* and z are constants. In both cases, the slope (z) of the linear functions can be interpreted as the rate of species accumulation as the (logarithm of the) sampled area increases.

The slope of SACs is influenced by key ecological properties of the sampled communities, such as richness, species abundances and spatial distribution (Flather, 1996; Cannon *et al.,* 1998). Furthermore, biotic (e.g. dispersal ability and reproductive behavior) and environmental (e.g. temperature and disturbance) factors, which drive species abundances and spatial distribution, may indirectly control this relationship (He & Legendre, 2002). Consequently, as anthropogenic disturbance, such as logging and fishing, alters communities through species depletion, geographical range contraction, local extirpation and habitat degradation (Dayton *et al.,* 1995; Ferretti *et al.,* 2010; Worm & Tittensor, 2011), SACs are expected to track this overall community changes. This is because as a community becomes less diverse, the rate of species accumulation with area decreases due to lower probability of finding species that are sensitive to exploitation or associated with undisturbed habitats, within a defined area. For instance, SACs applied to North American bird communities had progressively lower slopes as human use of land increased (Flather 1996). Tropical forests of West Kalimantan, Indonesia, and southeastern Madagascar showed less steep SACs in logged (exploited) versus unlogged areas (Cannon *et al.,* 1998; Brown & Gurevitch 2004).

Despite evidence of their utility in detecting human impacts on terrestrial communities, SACs are underutilized in marine systems, with few applications to fisheries (Neigel, 2003; Drakare *et al.,* 2006). These include studies that used SACs to estimate the species richness of a fish community (e.g. Ugland *et al.*, 2003), to examine the importance of scale and sample size when detecting the effect of fishing on benthic diversity (Kaiser, 2003), and to predict biodiversity losses in benthic communities at increasing habitat homogenisation (Thrush *et al.,* 2006). However, SACs have not been used as an index of human impacts on marine communities at large spatial and temporal scales. Among the several anthropogenic disturbances currently affecting benthic marine communities, bottom trawl fishing is considered one of the most severe and widespread (Jennings & Kaiser, 1998; Watling & Norse, 1998; Roberts, 2007). Trawling can have strong direct and indirect effects on marine communities by targeting a wide range of species, producing high levels of by-catch (Jennings & Kaiser, 1998) and impacting complex seafloor habitats (Dayton *et al.,* 1995; Thrush & Dayton, 2002).

In this study I aim to test SAC utility as an index of human impacts on demersal fish communities of continental shelves and slopes impacted by bottom trawling. Specifically, I analyzed data from bottom trawl surveys covering the whole South East Australia region and in some cases surveying pristine communities.

5.3 Methods

First, I defined my expectation on the effect of fishing on SAC. I used numerical simulations to investigate how changes in community properties (i.e. richness, species abundance and spatial distribution) alter the shape of SAC estimated from these communities. Then, I analyzed empirical data from bottom trawl surveys carried out in South East Australia over a period of about 20 years (1976 - 1997). These surveys sampled fish communities at different stages of exploitation and in some cases prior to fishing. Making use of these contrasts I: 1) explored trends in community properties along a fishing intensity gradient because, if fishing consistently changed community properties, a marked effect of fishing on SAC was expected; and 2) built SACs along spatial and temporal gradients of community exploitation to investigated whether the intensity of fishing was reflected in SAC parameter estimates.

5.3.1 Simulation-based analysis

To investigate the effect of community properties on z I simulated 40 communities, where ten differed in richness (R), ten in abundance (i.e. total number of individuals in a community, N), ten in evenness (i.e. how similar in abundance species of the same community are), and ten in intra-specific aggregation (i.e. how spatially clumped individuals of the same species are); I build a SAC for each simulated community and related changes in community properties to changes in z.

First I defined the rationale used to simulate each community. Given a sample of R species and N individuals, the initial step was to generate a vector of species relative abundance, characterized by a specific evenness. I used the zero truncated negative binomial distribution (TNBD) as data generation model, and assumed that the probability of any species R_i being represented by n individuals is:

$$q'_n = \frac{\Gamma(k+n)}{n!\Gamma(k)}\left(\frac{p}{1+p}\right)^n \frac{1}{(1+p)^k-1} \tag{5-3}$$

For $n = 1, 2, \ldots$

where k is the dispersion parameter and p is the scale parameter. The dispersion parameter, k, measures the shape of the species-abundance distribution (Pielou, 1975). Large values of k characterize species-abundance distributions with short tail indicating low proportion of rare species in the sample. Hence for large values of k evenness is high (Fig. 5-1). Further, k and p follow the relationship:

$$\frac{kp}{1-(1+p)^{-k}} = \frac{N}{R} \tag{5-4}$$

Where N/R is the observed mean of the TDNB (μ). By changing k (and the corresponding p), but holding μ constant we obtain vectors of species relative abundance differing in evenness (He & Legendre, 2002). I referred to k of the TNBD used to model evenness as k_b (i.e. dispersion parameter between species).

Given a vector of species relative abundance, defined by R, N and k_b, the next step was to simulate the spatial distribution of species over a matrix of $G_{100x100}$, where each grid cell (g_j) equals a meter squared. Species' spatial distribution needed to account for their degree of intra-specific aggregation. I used the negative binomial distribution (NBD) as data generation model, and I assumed that the probability of presence of R_i with abundance N_i in g_j is:

$$q'_n = 1 - \left(1 + \frac{\mu}{k}\right)^{-k} \qquad (5\text{-}5)$$

For $n = 1, 2, \ldots$

Where the dispersion parameter, k, reflects the degree of clustering between individuals (intra-specific aggregation) and μ is the average number of individuals per cell (N/G). Large values of k generate communities where the probability of R_i being found in a sample of area g_j is similar across all samples of area g_j. Hence, for large values of k intra-specific aggregation is low (Fig. 5-1). I referred to k of the NBD used to model intra-specific aggregation as k_w (i.e. dispersion parameter within species).

Next, I choose values of R, N, k_b and k_w used to parameterized eqn. 5-4 and 5-5, and ranging between those obtained from two bottom trawl surveys that sampled the same demersal fish community before and after exploitation. These surveys are later defined as strata 17 and 23 (Table 5-1). Bottom trawl surveys are sampling programs where fishing operations (tows), consisting in towing a cone shaped net in contact with the seabed, are performed over a designated area to collect biological information and indices of abundance of demersal fish (and invertebrate) species. Strata 17 sampled 22798 individuals belonging to 84 species, wherase strata 23 sampled 58568 individuals belonging to 97 species (i.e. R_{17}=84 and N_{17}=22798, R_{23}=97 and N_{23}=58568).

To estimate k_b, first I calculated the evenness index J' of Pielou (1975) for each stratum, then I inspected the relationship between k_b and J' (Fig. 10-1 in Appendix 3) and used it to extrapolate the comparative k_b ($J'_{17} = 0.71$, $k_{b\_17} \sim 0.5$; $J'_{23} = 0.58$, $k_{b\_23} \sim 0.3$).

To estimate k_w for each stratum, first I fitted a generalized linear model (GLM) with NBD to the catch data (number of individuals) of each species sampled in at least ten tows and considered the dispersion parameter of the NBD (k) a species-specific index of intra-specific aggregation (Pielou, 1977); then I averaged these indices. (See section 10-2 and Figs 10-2 and 10-3 in Appendix 3 for GLM model specifications, index of intra-specific aggregation, and GLM model diagnostics, respectively). However, k_w ranging between values reported for strata 17 and 23 ($0.67 \leq k_w \geq 0.87$) did not generate communities significantly differing in intra-specific aggregation (this was graphically tested), thus I chose a wider range of k_w ($0.01 \leq k_w \geq 1.5$). (See Table 10-1 in Appendix 3 for R, N, k_b and k_w values used for simulations).

Last, for each simulated community I randomly sampled 100 cells out of G and used data from these cells to build SACs.

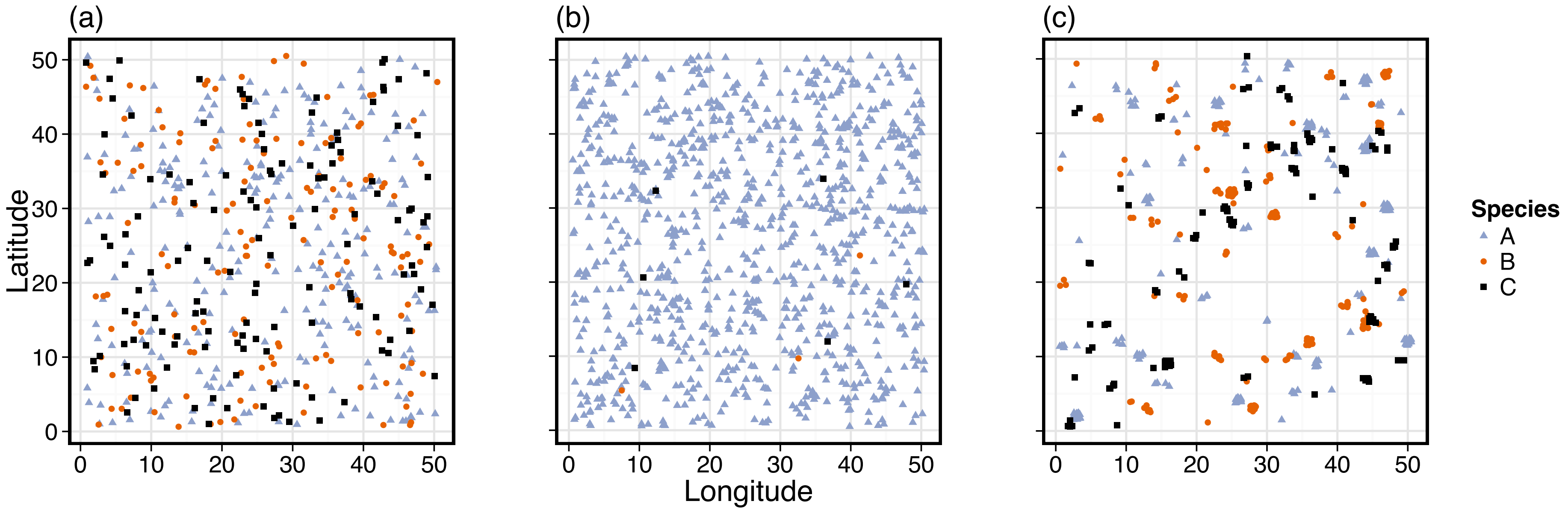

Figure 5-1. Species distribution over a matrix of G_{50x50}. Parameters used: (a) R=50, N=1000, k_b=1 and k_w=1 (i.e. high evenness and low intra-specific aggregation); (b) R=50, N=1000, k_b=0.01 and k_w=1 (i.e. low evenness and intra-specific aggregation); and (c) R=50, N=1000, k_b=1 and k_w=0.01(i.e. high evenness and intra-specific aggregation).

Table 5-1. Strata details. *CCTE* is the Cumulative Commercial Trawling Effort index. *Swept area*, *Sam. area* (*Sampled area*) and *Survey area* are spatial properties of each stratum. *Headrope* and *Code-end* are net characteristics. For each stratum I considered the net with the highest percentage on use, in parenthesis.

Agency	Strata	Year range	Mean depth (m)	Mean lat. DD	*Swept area* (km^2)	*Samp. area* (km^2)	*Survey area* (km^2)	*CCTE* (h/km^2)	*Headrope; Code-end* (%)
NSW DPI	1	77-78	134	-32.7	0.06	3	6521	1.4	21;90 (100)
-	2	83-84	1018	-35	0.14	6	649	0	21;90 (77)
-	3	83-84	1042	-33.5	0.17	7	1672	0	21;90 (51)
-	4	87-89	1062	-35.2	0.11	7	788	6.4	30;90 (71)
-	5	87-89	1066	-33.1	0.11	10	2106	0.5	30;90 (60)
-	6	93-94	48	-35.2	0.16	10	21	12.3	56;42 (100)
-	7	93-94	144	-34.4	0.16	10	103	41.3	56;42(100)
-	8	92-94	122	-35.7	0.16	11	64	86.1	56;42 (100)
-	9	92-94	126	-33.1	0.16	11	64	0	56;42 (100)
-	10	92-94	142	-32.8	0.16	13	1283	2.9	56;42 (100)
-	11	92-94	34	-33	0.16	12	26	1.1	56;42 (100)
-	12	93-94	118	-36.6	0.16	10	32	5.1	56;42 (100)
-	13	93-94	147	-37.3	0.16	10	213	19.4	56;42 (100)
-	14	93-94	46	-36.7	0.16	10	22	8.7	56;42 (100)
-	15	76-77	405	-35.6	0.06	6	117	0	21;90 (100)
-	16	76-77	405	-33.6	0.07	6	545	0	21;90 (100)
-	17	76-77	405	-37.6	0.06	4	953	0	21;90 (100)
-	18	79-81	415	-35.4	0.19	14	1303	0	56;90 (90)
-	19	79-81	425	-33.3	0.2	8	949	0	56;90 (84)
-	20	79-81	386	-37.7	0.15	12	1087	0	56;90 (78)
-	21	96-97	414	-35.6	0.06	3	143	41.3	21;90 (100)
-	22	96-97	405	-33.6	0.06	3	414	14.4	21;90 (100)
-	23	96-97	408	-37.5	0.06	4	345	33.9	21;90 (100)
IMAS	24	79-82	111	-41.8	0.17	16	13055	1.2	38;90 (35)
-	25	79-82	47	-40.5	0.1	2	18955	0	29;90 (45)
-	26	79-82	152	-43.4	0.18	9	9963	0.2	38;90 (71)
-	27	79-82	578	-41.4	0.26	14	3878	0.2	60;90 (37)
-	28	83-87	124	-43.5	0.16	4	5079	0.6	34;110 (48)
-	29	1983	950	-41.5	0.37	18	3863	0.3	53;90 (49)
-	30	93-95	82	-42.7	0.04	4	1025	1.6	26;20 (100)
-	31	93-95	50	-43.2	0.04	4	911	2.3	26;20 (100)
CSIRO	32	1988	940	-35.5	0.1	2	24	5.4	35;37 (100)
-	33	1984	446	-42	0.06	3	331	0	35;37 (97)
-	34	88-89	975	-40.6	0.06	3	3128	2.3	35;37 (100)
-	35	88-89	972	-41.6	0.07	7	3914	2.9	35;37 (100)

5.3.2 Calculating SACs for simulated communities

To build SAC I used the species sampled and the area per grid cell g_j as a sample unit. I drew a series of n cells, where n goes from 1 to the total number of cell sampled, and progressively accumulated the number of species found in each cell group, as well as the cell area. To consider different possible combination of cells in each draw, I repeated the process 100 times, and averaged the series of cumulative species and cell areas obtained. Lastly, I fitted eqn. 5-2 to each SAC and estimated z for each community simulated. I chose eqn. 5-2 because it was the function best describing SACs derived from empirical survey data (see section 'Calculating SAC for strata').

5.3.3 Scientific bottom trawl surveys

I used trawl survey data provided by three research agencies: 1) the New South Wales Department of Primary Industries (NSW DPI); 2) the Institute for Marine and Antarctic Studies in Tasmania (IMAS); and 3) the Commonwealth Scientific and Industrial Research Organization (CSIRO). From each agency I obtained a dataset of trawl tows information including latitude, longitude, date and depth of the tow, and fish species caught in each tow. As most of the surveys' catch data consisted of demersal bony fishes and elasmobranchs, I restricted our analyses only on these organisms.

The trawl surveys differed in spatio-temporal coverage; therefore I stratified the survey dataset into 35 relatively homogeneous spatio-temporal strata. Each stratum identified a specific geographic area, depth range (i.e. continental shelf [10-200 m], upper continental slope [200-650 m], and mid-slope [>650 m]) and sampling period (see Fig. 5-2, Table 5-1, section 10-1 and Table 10-2 in Appendix 3 for map of surveys' tows, strata details, datasets description, and sampling specifics, respectively.)

The NSW DPI provided the set of strata with the greatest sampling resolution (strata 1 to 23). These strata reported the number of individuals sampled per species for all strata' tows, whereas data from other agencies often lacked this information. While I applied SACs to all strata considered (1 to 35), strata 1 to 23 allowed me to explore trends in community properties along a fishing intensity gradient. Furthermore, strata 15 to 17 surveyed communities of the continental upper slope of New South Wales, in 1976-77, before any trawling took place, and strata 21 to 23 surveyed the same communities after 20 years of trawling, thus representing the best case of sampling before and after exploitation.

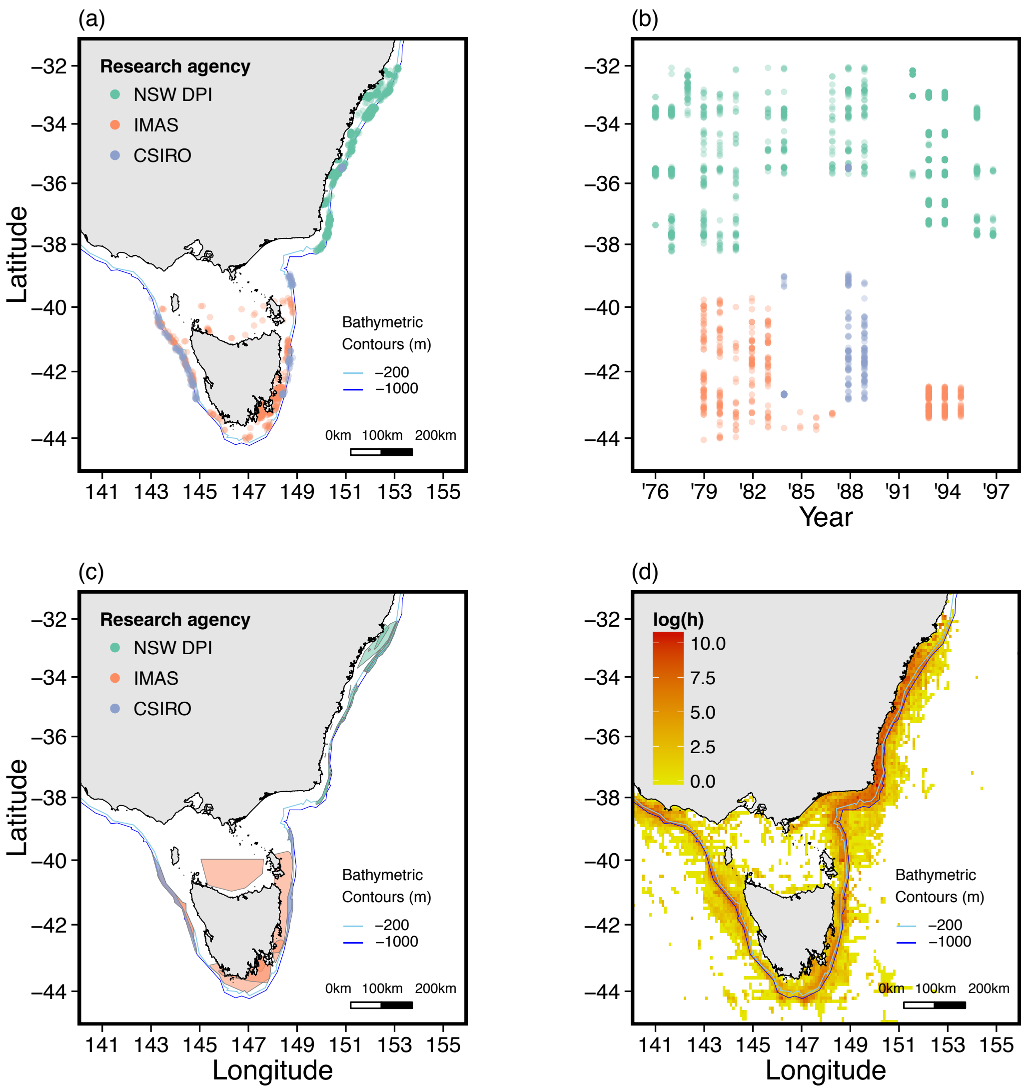

Figure 5-2. Distribution of data: (a) surveys' tows locations; (b) Latitudinal (in bins of 0.1 DD) and temporal coverage of surveys' tows; (c) polygons representing surveys' strata; and (d) commercial trawling intensity along the coast of South East Australia, for the period 1918-1997. [2]

[2] Acronyms used in Fig. 5-2: NSW DPI = New South Wales Department of Primary Industries; IMAS = Institute for Marine and Antarctic Studies; and CSIRO = Commonwealth Scientific and Industrial Research Organization.

5.3.4 Strata features

To account for sampling differences across strata I characterized each stratum with the spatial scale covered during sampling and the trawl net used.

I calculated the *Swept area* (mean swept area per tow), the *Sampled area* (sum of all tows' swept area), and the *Survey area* (area of the smallest polygon enclosing all the stratum tows). Strata with a wider *Swept*, *Sampled*, and *Survey area* are expected to accumulate more species because of a greater chance of sampling different habitats, and communities farther from one another are likely to have less species in common (e.g. Palmer & White, 1994).

I considered the *Head-rope* length (size of the trawl net at the mouth), and the *Code-end* mesh size (mesh size of the last section of the trawl net). Differences in these net's features may determine differences in fishing efficiency and selectivity (e.g. Reeves *et al.,* 1992), thus number of species and individuals sampled. As occasionally a combination of nets was used to sample the same stratum, for each stratum I considered *Head-rope* and *Code-end* of the net with the highest percentage of use (Table 5-1 and Table 10-2 in Appendix 3).

To characterize the sampled communities for the level of fishing exploitation they sustained, for each stratum I estimated an index of cumulative fishing effort since the onset of exploitation, here referred to as Cumulative Commercial Trawling Effort index (*CCTE*) (Table 5-1). That was the sum of hours trawled (h) in the *Survey area* from the first commercial tow to the time the stratum was surveyed, per unit area (km^2). Data on commercial trawling operations were extracted from commercial logbooks available for the period 1918-1957 (Klaer, 2006) and for the period 1974-

1997 (obtained from the Australian Fisheries Management Authority, AFMA). (See Fig. 5-2d for commercial trawling intensity along the coast of South East Australia.)

5.3.5 Changes in community properties

I calculated richness (*R*), abundance (*N*), evenness (*J'*) and degree of intra-specific aggregation (k_w) for strata 1 to 23 and related them to *CCTE*. This was to test whether community structure changed along a fishing intensity gradient. To estimate k_w I followed the approach used for strata 17 and 23 in section 'Simulation-based analysis' and fully described in section 10-2, Appendix 3.

5.3.6 Calculating SACs for strata

I built a SAC for each stratum. I followed the approach used to calculate SACs for simulated communities, but considered each stratum's tow (tow_i) as a sample unit. I explored the performance of eqns 5-1 and 5-2, and several other common functions for SAC reported in the literature (Flather, 1996; Tjørve, 2003). As model selection indicated that eqn. 5-2 was the best function in 29 out of the 35 strata, I used eqn. 5-2 for all my steps. (See section 10-3 and Table 10-3 in Appendix 3 for SAC functions and selection.)

5.3.7 Statistical analysis of SACs

I fitted linear mixed affects (LME) models to the entire dataset of SACs calculated for each stratum to quantify the effect of trawling intensity on *z*. My model was defined as

$$S_{ij} = c + z * log\ (A)_{ij} + \beta * X_{ij} + b_i + \varepsilon_{ij} \qquad (5\text{-}6)$$

$b_i \sim N(0, \sigma_b^2);$

$\varepsilon_{ij} \sim N(0, \rho\, \varepsilon_{ij-1} + \eta_{ij})$

$\varepsilon_{ij} \sim N(0, \sigma^2 * |A_{ij}|^{2\delta})$

For $i = 1,\ldots,35$

For $j = 1,\ldots,n$

where S is the cumulative number of species and $log(A)$ is the log-transformed cumulative area (of series of n tows); both are output of SAC calculated for each stratum. The dependent variable, S, originated from an average of 100 SAC calculations; it is given as decimal and modeled assuming a normal distribution. The parameters c and z are SAC intercept and slope, respectively; X is a matrix of covariates. The term b_I refer to the random intercept (stratum), and the indices i and j refer to stratum and observations within a stratum, respectively. Lastly, the term ε_{ij} is the within stratum variation. I included a first order autoregressive correlation structure to account for correlation between residuals (ε_{ij}) at $log(A)_{ij\,-1}$ and residuals at $log(A)_{ij}$, and a power variance structure to account for decreases in residuals spread for larger A.

The matrix of covariates, X, included the *CCTE* index per stratum and a set of other covariates defined on the basis of ecological and sampling considerations. In particular, to account for differences in community structure due to environmental heterogeneity, I considered stratum mean depth (*Depth*) and mean latitude (*Latitude*). Additionally, to account for sampling differences between strata surveyed by different research agencies I considered the covariate *Agency,* of three levels (NSW DPI, IMAS and CSIRO), and to account for more specific sampling differences across strata I considered *Swept area*, *Sampled area*, *Survey area*, *Headrope*, and *Code-end* (Table 1). However, *Swept area*, *Sampled area,* and *Headrope* were positive correlated, thus I discharged the variables *Swept area* and *Headrope.*

In the LME model, linear terms gave information on covariates' effect on *c,* whereas first-order interactions between *log(A)* and the other covariates influenced the actual slope of the SAC.

Lastly, I fitted eqn. 5-6 but considered a different specification of the fishing effort index ($CCTE_{10y}$) and compared models' results. Whereas *CCTE* was calculated considering all commercial tow (from 1918), I calculated $CCTE_{10y}$ as the sum of hours trawled in the *Survey area* during the ten years before the stratum was surveyed, per unit area. This was to test whether fishing exploitation was mostly confined to more recent years.

All analyses carried out in this study were implemented in R 3.1.0 (R Development Core Team, 2014).

5.4 Results

5.4.1 Simulation-based analysis

Numerical simulations showed that *z* is sensitive to changes in all community properties we considered (Fig. 5-3), particularly richness, abundance and evenness. Low values of *z* were encountered when communities were characterized by low richness (*R*=84), abundances (*N*=22798) or evenness (*J'*=0.58). Species spatial aggregation marginally affected *z*, which decreased as intra-specific aggregation decreased, but had a notable effect on its sampling error (i.e. for high values of intra-specific aggregation, z had large confidence intervals; k_w=0.01; z=14±1.2). This is expected when aggregation of individuals increases, as there is a greater proportion of the total area with no individuals, and sampling has a lower detection frequency (Fig. 5-1). Thus, estimates of z depend on how frequently we randomly sample areas with no individuals.

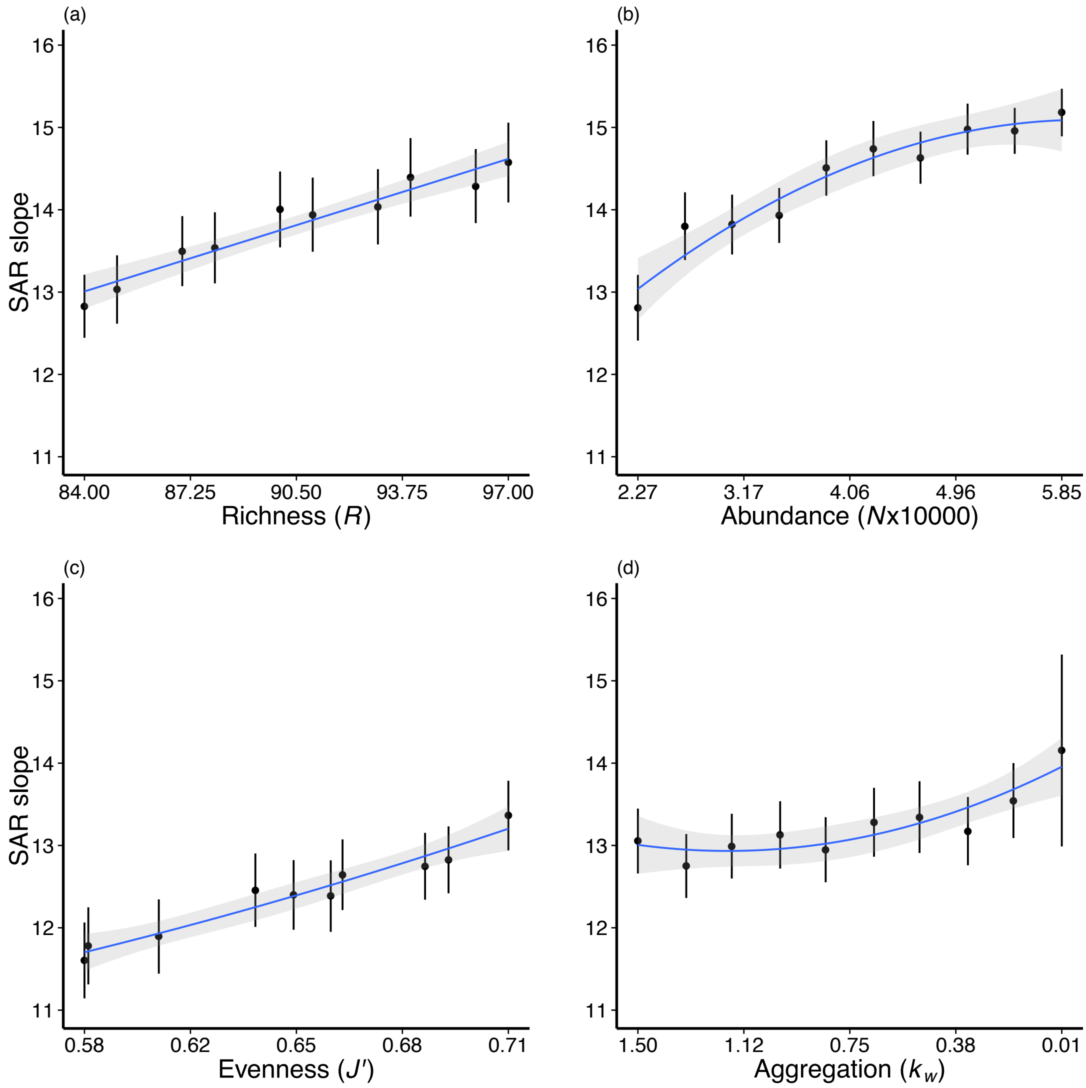

Figure 5-3. Results from numerical simulations showing the effect of (a) richness, (b) abundance, (c) evenness and (d) intra-specific aggregation on z.

5.4.2 Changes in community properties

Community properties changed along trawling intensity gradients (Fig. 5-4). On the continental shelf, richness and evenness decreased at increasing *CCTE,* whereas intra-specific aggregation increased. On the upper continental slope, richness and abundance remained constant; evenness was highly variable and intra-specific aggregation decreased. On the mid-slope, all properties remained fairly constant.

Overall, the direction of change along an increasing gradient of fishing intensity was inconsistent across community properties, thus there was no clear expectation on how *CCTE* should have affected SAC describing these communities.

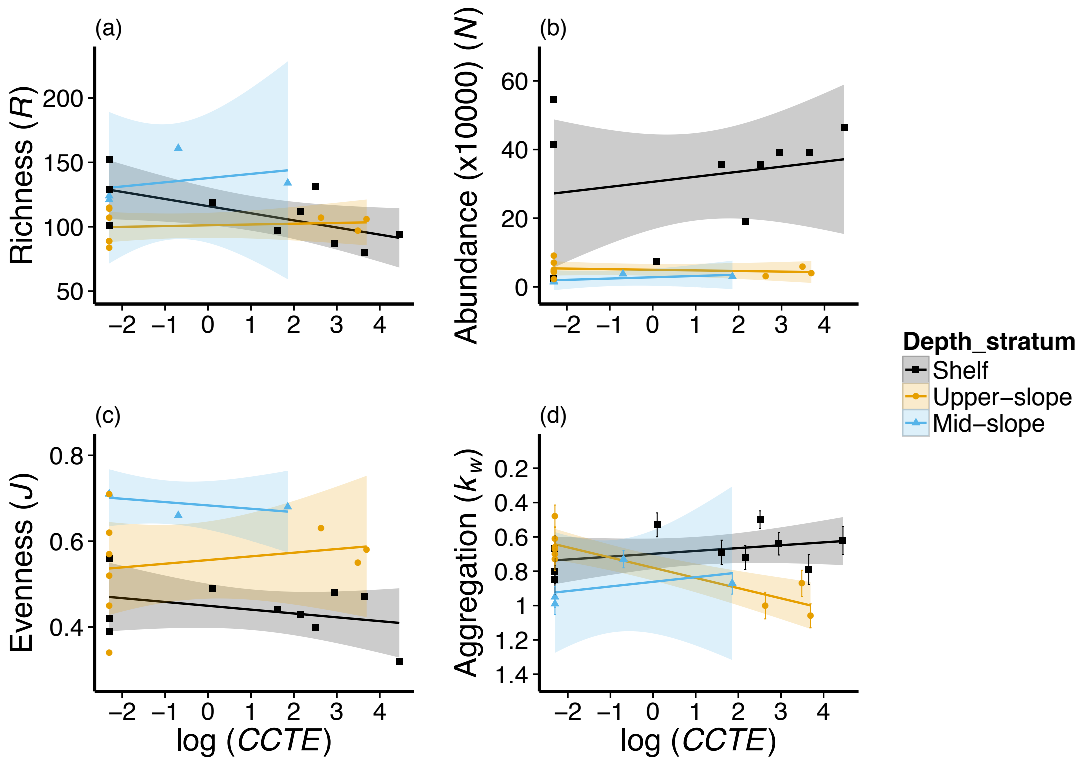

Figure 5-4. Richness (a), abundance (b), evenness (c) and degree of intra-specific aggregation (d) along depth and Cumulative Commercial Trawling Effort (*CCTE*) (h/km^2) gradients. In (d) standard errors for stratum-specific values of k_w are shown.

5.4.3 Variability in SAC across strata

SACs showed variation in shape across strata (Fig. 5-5). The mean value of *z* across all strata was 19.2 with a range from 9.2 to 37 (Table 5-2). Although there seemed to be little difference in SAC shape as *CCTE* increases (Fig. 5-5), summaries of *z* at low (zero), medium (0.1 to 20) and high (>20) levels of *CCTE* did show some variation (Table 2). The slope decreased (flatten the SAC) as *CCTE* increases, but there was a greater variability of *z* across strata than along the gradient of *CCTE* (Fig. 10-4 in Appendix 3).

In contrast, comparison between strata 15 to 17 and 21 to 23 revealed an increase in *z* after 20 years of trawling (Fig. 10-4 in Appendix 3). The mild increase in *z* between strata 15 and 21, and strata 16 and 22 were due to relatively high richness despite much lower *Sampled area* for strata 21 and 22. The marked increase in *z* between strata 17 and 23 (z_{17}=15.2; z_{23}=20.2) was due to an increase in richness and abundance at equal *Sampled area*, and despite decreases in evenness and intra-specific aggregation. On one hand this highlighted the importance of accounting for differences in spatial scale across strata, on the other hand it showed unexpected (i.e. increases) and antagonistic changes in community properties after fishing. (See Table 5-1 and Table 10-4 in Appendix 3 for strata description and community properties, respectively.)

Table 5-2. Summaries of *z* across all strata, and at low (zero), medium (0.1 to 20) and high (>20) levels of Cumulative Commercial Trawling Effort (*CCTE*) (h/km^2). Standard Deviation (SD) is reported for mean *z* values.

CCTE	*z* mean	SD	*z* min	*z* max
All	19.2	5.6	9.2	37
Low	19.8	4.1	13.9	26.1
Medium	19.1	6.7	9.2	37
High	18.1	4.4	13.1	23

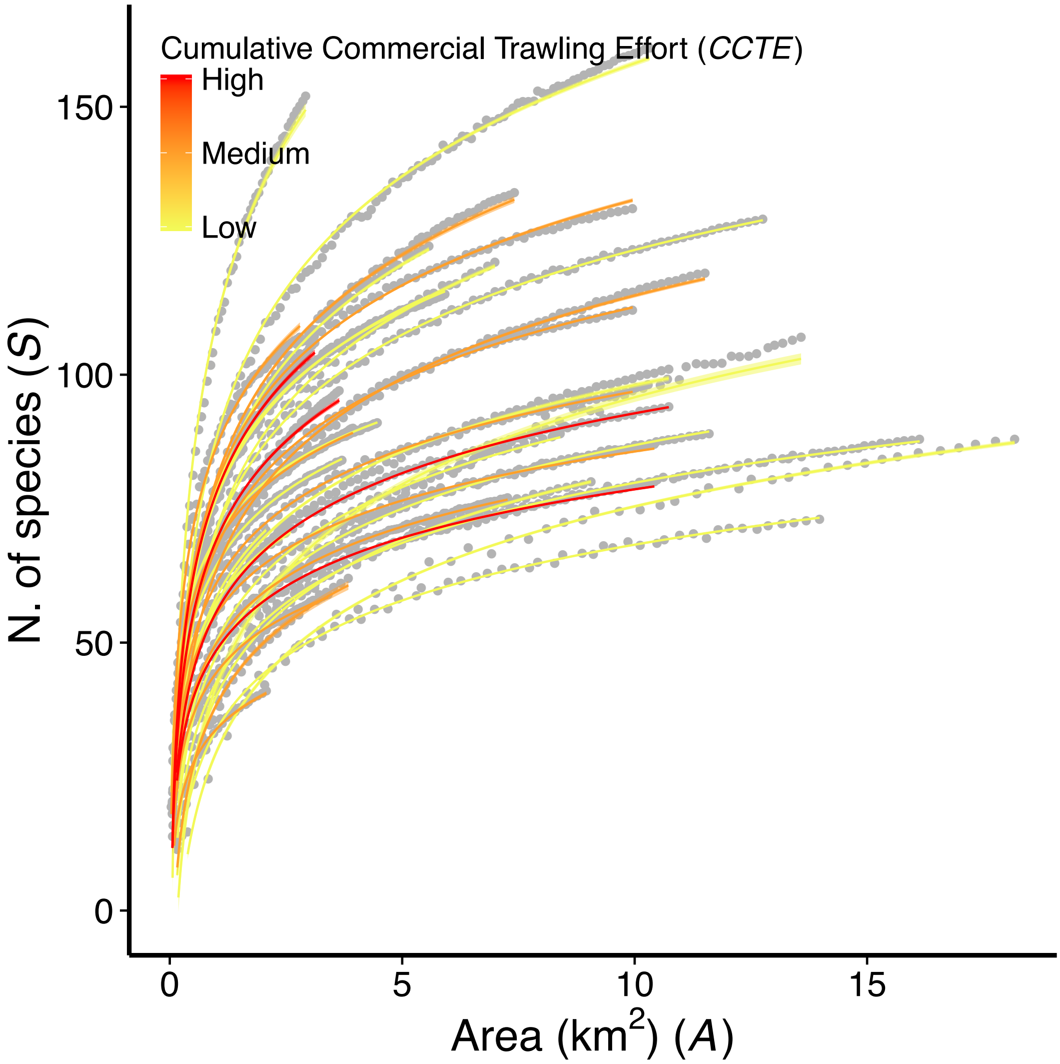

Figure 5-5. Empirical SAC calculated for each stratum (dots), and exponential function fitted to each SAC (smooth lines). Colors indicate the intensity of Cumulative Commercial Trawling Effort (*CCTE*) (h/km^2) characterizing each stratum ("Low" when *CCTE* equals zero; "Medium" when *CCTE* is between zero and 20; "High" when *CCTE* is greater than 20).

5.4.4 Statistical analysis of SACs

The LME model showed that *CCTE, Depth, Latitude, Agency, Sampled area* and *Survey area* had a significant effect on SAC slopes (Table 5-3 and Fig. 5-6). In particular, *CCTE* had a negative effect on *z*, indicating that SACs estimated from exploited demersal fish communities were less steep than those from less exploited regions (Fig. 5-6). Results also show that *Depth* and *Latitude* had a strong positive effect on *z*, with *z* markedly increasing for deeper communities and communities closer to the tropics. The magnitude of these effects implied that *z* chiefly depends on environmental drivers. *Agency* influenced *z*, with strata surveyed by CSIRO showing remarkably lower *z* than strata surveyed by NSW DPI and IMAS, possibly because of differences in survey's focus and sampling. *Sampled area* and *Survey area* had an opposite effect on *z.* Whereas increases in *Sampled area* lowered *z*, increases in *Survey area* increased *z*. The effect of these covariates emphasized the importance of accounting for spatial scale when comparing SACs. However, because the covariate *Sampled area* was correlated with *Swept area* and *Headrope*, thus captured a combination of strata characteristics, its effect on *z* may be biased. In contrast, there was no evidence of a significant effect of *Code-end* on *z*. (See Fig. 10-5 in Appendix 3 for model diagnostics.)

Model results did not change when $CCTE_{10y}$ (not shown here) was used instead of *CCTE,* suggesting that most of the fishing effort captured by the two indices was confined to years closed to sampling.

Table 5-3. Results from Linear Mixed Effects (LME) model. *CCTE* refers to the Cumulative Commercial Trawling Effort (h/km^2), and SE is the Standard Error. *log(A)*, *Sampled area* and *Survey area* are in km^2; *Depth* is in meters, *Latitude* is in Decimal Degrees (DD) and *Code-end* is in mm.

Parameter	Variance	SD
Intercept	64.24	8.02
Residual	1.11	1.05

Covariate	Estimate	SE	t	p-value
Intercept	63.98	6.28	10.18	<0.001
CCTE	-3.22	1.56	-2.06	0.049
Depth	8.17	2.15	3.81	0.001
Latitude	2.07	3.3	0.63	0.537
Agency NSWDPI	27.2	8.78	3.1	0.005
Agency CSIRO	-34.8	8.14	-4.28	<0.001
Sampled area	-9.9	1.52	-6.52	<0.001
Survey area	5.24	1.93	2.71	0.012
Code-end	-8.46	2.39	-3.55	0.002
log(A)	17.72	0.76	23.33	<0.001
CCTE:log(A)	**-0.86**	**0.17**	**-5.18**	**<0.001**
Depth:log(A)	1.91	0.18	10.46	<0.001
Latitude:log(A)	1.83	0.39	4.64	<0.001
Agency NSWDPI:log(A)	2.91	1.02	2.84	0.004
Agency CSIRO:log(A)	-7.52	0.95	-7.95	<0.001
Sampled area:log(A)	-0.65	0.18	-3.62	<0.001
Survey area:log(A)	1.22	0.22	5.67	<0.001

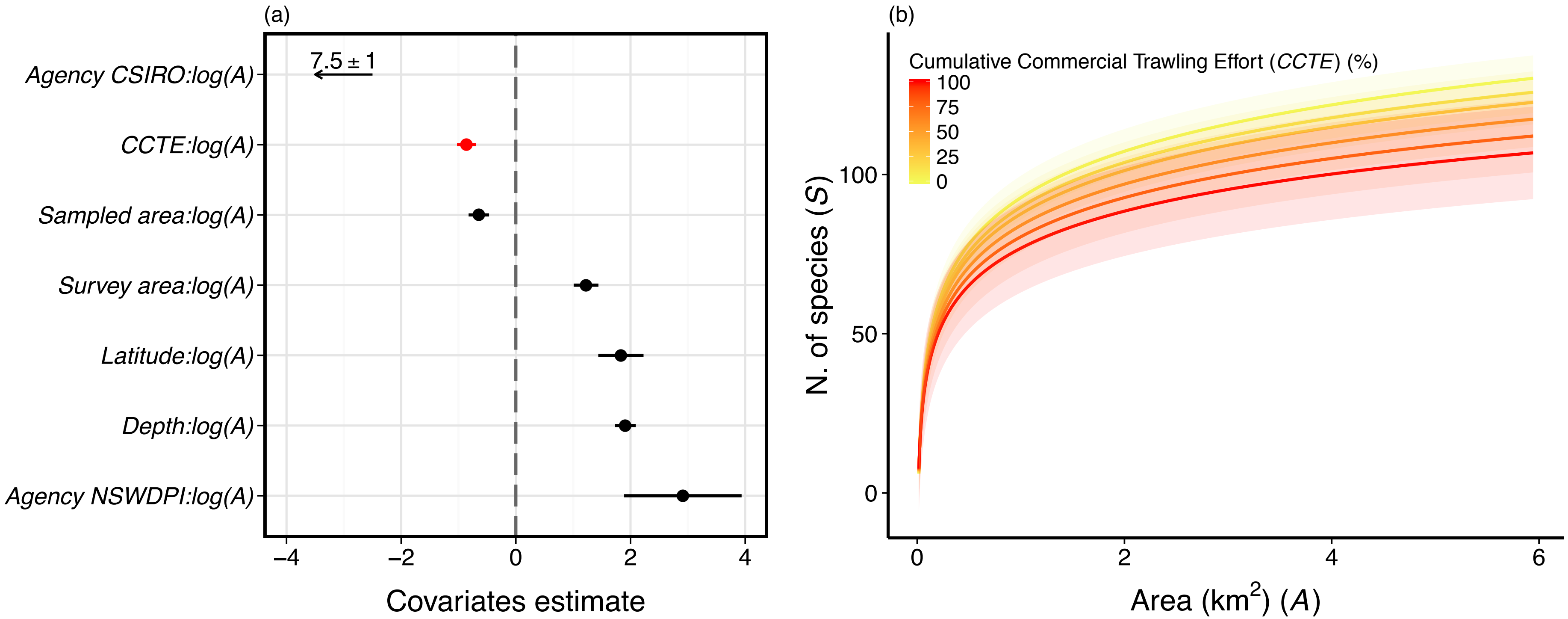

Figure 5-6. Results from mixed effects (LME) model. (a) Estimates of interaction term coefficients and confidence intervals (CI); and (b) SAC model prediction at increasing levels of *CCTE*, where 0% indicates pristine communities, and 100% indicates communities subjected to the maximum level of *CCTE* observed across strata.

5.5 Discussion

This study highlighted that species accumulation curves can be used as an index of human impact on marine communities. I used a mix of numerical simulations and an extensive dataset of historical trawl surveys carried out in Australia to test species accumulation curves as tools to reveal fish community depletion. Numerical simulations demonstrated that the shape of these curves is affected by community properties such as richness, abundance, evenness and intra-specific aggregation. A decrease in any of these properties decreased SAC slope (Fig. 5-3). Analyses on the trawl survey data showed that SAC slope was able to capture differences in community structure due to environmental variation along a latitudinal and depth gradient, and it was sensitive to sampling. After controlling for these factors, I found that SAC slope decreased proportionally to the amount of commercial trawling that had been deployed in each community since the onset of exploitation (Fig. 5-6). This result suggested that demersal fish communities of South East Australia have undergone structural change since the industrialization of fishing in the region and that SACs are instrumental to detect such change.

In South East Australia trawling has been linked to the decline of target and by-catch species abundances on continental shelves (e.g. Klaer, 2001) and slopes (e.g. Andrew *et al.,* 1997; Graham *et al.,* 2001; Tuck, 2011). These changes have occurred because of the direct and unselective removal of fish and invertebrates from the community and because of physical disturbance on sediments and structured habitats by trawl nets (e.g. Andrew *et al.,* 1997; Watling & Norse, 1998). Elsewhere in the world, communities under intense trawl exploitation were characterized by less diverse habitats (and associated species) (Dayton *et al.,* 1995; Collie *et al.,* 1997), thus expected to show less steep SAC slopes (Thrush *et al.,* 2006).

While the effect of fishing can reduce SAC slope, the effect size was small in relation to the one given by environmental variability (expressed by the effect of the covariates *depth* and *latitude*). This pattern suggested that the structure of demersal fish communities is greatly influenced by environmental drivers. Variation in demersal fish community structure according to depth and latitude is widely documented (e.g. Williams & Bax, 2001), with community richness generally increasing from poles to tropics (Willig *et al.,* 2003) and from the shallow continental shelf to the deeper slope (McClatchie *et al.,* 1997; Sousa *et al.,* 2006), although exceptions do exist (Gray, 2001). Spatial variation in demersal fish community structure is region-specific and may be associated with a range of factors, such as hydrography and seabed type (e.g. Williams & Bax, 2001). This aspect has important implications for comparing SAC from different marine regions across large spatial scale. For instance, comparing SAC slopes derived from communities across ocean regions that greatly differ in environmental features may fail to inform on these communities' status of depletion. This is because the effect of exploitation may be difficult to disentangle from that of environmental variation. Instead, a region-specific rate of change in SAC slopes can be obtained if, for each region, survey data are available across a gradient of fishing intensity. Region-specific rates of change may then be compared to determine the relative magnitude of these communities' depletion.

On one hand, the comparatively low effect of exploitation on SAC slope may be due to a limited impact of trawling on demersal fish communities of South East Australia.

In South East Australia the history of trawl fishing is fairly short. Trawling started in 1915, but was confined to the continental shelf of New South Wales until the mid 1970s, when it expanded to deeper grounds and Tasmanian regions (Tilzey & Rowling, 2001). Even then, possibly because Australian waters are less productive than other continental shelves (Molony *et al.,* 2011), trawling intensity has never reached the levels characterising some of the most productive fisheries around the world, such as the North Sea (Jennings *et al.,* 1999; Larcombe *et al.,* 2001). For example in 1995, toward the end of our observation period, mean trawling intensity in South East Australia was about 0.45 hours/km^2, less than a quarter of the mean trawling intensity reported for the same year in the North Sea (~ 1.73 hours/km^2; [Jennings *et al.,* 1999]).

On the other hand, the low effect of exploitation on SAC slope may be due to the inability of the fishing effort index (*CCTE*) to captured the true extend of communities depletion. *CCTE* only reflects trawling intensity, and it is unclear whether this is a good proxy of impact on fish communities; i.e. community depletion may not be directly proportional to the amount of trawling deployed. Trawling may have the greatest impact on fish communities in an initial phase of exploitation (Ward & Myers, 2005). Then it can continue to deteriorate the fish communities at a slower rate as exploitation proceeds. For most of the communities considered, however, trawling exploitation was mostly concentrated in the ten years before sampling, thus sampling captured either the first or the most intense phase of exploitation. Therefore we expected that *CCTE* was proportional to the level of community depletion. While other fisheries, pollution and climate change have contributed to alter the communities of South East Australia (Johnson *et al.,* 2011), trawling has been the major fishery, and most likely the main source of impacts (Tilzey & Rowling, 2001;

Woodhams *et al.,* 2011). Thus *CCTE* might reflect most of the intensity of anthropogenic disturbance.

Lastly, SAC may have failed to capture some of the trawling impacts on demersal fish communities because SAC sensitivity to changes in community structure may be reduced if the effects of community properties on SAC slope are antagonistic. An extreme case is that of strata 17 and 23, where increases in observed richness and abundance overcome decreases in evenness and intra-specific aggregation, so that SAC slope increased after trawling (Fig. 10-4 and Table 10-4 in Appendix 3). Antagonistic changes in community properties were reported for other communities. For example, unfished benthic communities of Georges Bank were characterised by higher richness and abundance but lower evenness than fished sites (Collie *et al.,* 1997). Although in such cases SAC may not capture all effects of fishing on communities, if used to compare diverse communities it can detect overall patterns.

Detecting community depletion with SAC has great potential. Species accumulation curves can express the effects of fishing on fish communities with a single index, the SAC slope, which requires the minimal amount of information. Bottom trawl surveys are available from virtually any continental shelf and slope of the ocean (e.g. Worm *et al.,* 2009), and, although the resolution and details of these surveys can vary even drastically across and within surveys (a reason for which many historical surveys remain underutilized, Ferretti *et al.,* 2014), the list of species detected for each tow and the associated area swept is available in most of occasions. SACs provide a summary of communities' exploitation status more efficiently than using single species or community composition approaches (e.g. cluster analyses), especially if the communities under investigation are many and the goal is to flag depletion rather than to explore changes at fine-scale community structure. To understand global change due to human impact synthetic indices of communities' health (Halpern *et al.,* 2012) are increasingly needed. For instance, region-wide comparison of SAC slopes along gradients of human exploitation can inform on demersal fish communities' status (e.g. depleted, stable, recovering) thus aid communities' monitoring. Further, across-regions comparisons of rate and direction of changes in SAC slopes can be used to rank demersal fish communities for their level of depletion. This might result in a better understanding of the distribution and magnitude of impacts, with important consequences for conservation prioritization.

6 Chapter 6 – Conclusion

This thesis examined the long-term effect of fishing on demersal fish communities of South East Australia and has highlighted the importance of historical perspective and retrospective records to comprehend and track the full extent of ecological changes on natural communities. By reviewing the history of fishing exploitation, management and research in South East Australia I identified trawling as one of the main sources of fishing impact on marine ecosystems of the region and I selected scientific bottom trawl surveys carried out between 1898 and 1997 as the best available data that could inform on changes in fish populations and communities along temporal gradients of fishing intensity (chapter 2). Due to the temporal span of the data, its origins and formats, I inevitably faced challenges. These related to the gathering of all the relevant and available information, its standardization (chapter 3) and the need to identify and develop methods of analysis that overcame data gaps and limits. Despite these challenges, I showed that the resulting long-term dataset was able to clarify causes and rates of change for demersal fish communities. In particular, I compared pre- with post-trawling exploitation data (1898-1910 and 1980s-2010s, respectively) and found marked changes in some components of the demersal fish communities and steep declines in many key commercial fishes (chapter 4). These changes were most likely related to the effect of fishing. Next, I tested the use of species accumulation curves applied to the survey dataset to determine their ability to capture trawling-induced changes in demersal fish communities (chapter 5). I found that the rate of species accumulation with area (the slope of species accumulation curves) decreased at increasing trawling exploitation. This indicated that trawling modified community structure through the removal of particular species, and through changes in the abundance and spatial heterogeneity of those remaining (chapter 5).

6.1 Implications of the findings

This work has delineated practical and analytical challenges that are likely to be common to the collection, standardization and analysis of most historical data (Klaer, 2006; Novaglio, 2010; Ferretti *et al.,* 2014), and the framework I adopted may provide guidance in similar situations. This followed three main steps. First is a review of the history of fishing exploitation to identify potential sources of impacts on marine communities, and historical data that could be informative about long-term changes to these communities. Secondly, the available data need to be gathered and standardized to a format ready for analysis. This step involves a considerable amount of effort and should not be underestimated in allocation of time and resources. The standardization (e.g. adjusting for missing tow position and net characteristics, obtaining a common measure of sampling effort and updating species names) requires the use of multiple sources of information, including surveys' reports and biological records, old literature, and experts' opinion (e.g. taxonomists and fishery scientists can advise on old scientific names and net's specifications). Thirdly, the use of new analytical methods that overcome data gaps and limits to reconstruct the baseline (pre-fishing) structure of marine communities and to clarify causes and rates of ecological changes.

Steps 1 and 2 were a fundamental part of the work and the resulting dataset is itself a main product. This dataset was used to answer important questions in the present thesis, and is available for further interrogation (it will be available on-line). The strengths of the dataset are that it spans the full temporal extent of trawling exploitation in South East Australia and that it includes information on the whole demersal fish community, rather than on the main target species exclusively. Both are rare characteristics. This is because the collection of fishery data usually begins at some time after the start of exploitation, thus missing the first stage of impacts. Additionally, most long-term datasets on fish abundance are derived from the fishing industry as logbook records (e.g. Klaer, 2001) or from Government agencies as fishery statistics (e.g. Thurstan *et al.,* 2010), and thus focus exclusively on species of commercial interest and value.

I provided new approaches to analysis that overcame data gaps and limits and that can be applied in other similar circumstances, particularly where data are patchy and of variable quality across sampling programs. In chapter 4 I dealt with differences in sampling design of surveys widely spaced in time by selecting overlapping areas, similar sampling gears, and an equal number of sampling operations between the periods considered. Also, I coped with differences in taxonomic resolution by adopting the broadest classification available, and I calculated indices of abundance using the most consistent information across periods (i.e. either presence or abundances as number of individuals). When I had enough data (i.e. for gurnards, flatheads and morwongs) I calculated standardized index of abundances before and after exploitation using Generalized Linear Models (GLMs) that accommodate unbalanced sampling designs. Although often having to sacrifice data resolution, I was still able to detect patterns of change. In chapter 5 I adopted analytical techniques that required minimum information yet proved to be sensitive to changes in community properties (richness, abundance and spatial distribution of species) and able to track fishing (trawling)-induced changes in community structure. Species Accumulation Curves are not new in the ecological literature, but their application as index of human impacts on demersal fish communities using long-term bottom trawl survey data is.

The information provided in this study can have direct application to assessment and management of the resource. For example, results from chapter 4 can be used to review and improve stock assessment for flatheads and morwongs. This is particularly the case for the Tasmanian component of the assessment, where data from 1985 are used as the baseline (Klaer, 2010; Wayte & Fay, 2011) although my results indicated previous steep decreases in the abundances of these families. Also, I identified families, such as gurnards, that are not currently assessed because of relatively low commercial value, but that have clearly been subjected to a marked decline since exploitation began, thus deserving further consideration. In addition, species accumulation curves, examined in chapter 5, provided a synthetic index of community health that tells us much about demersal fish community status (e.g. depleted, stable, recovering). Such indicators can assist in monitoring the impacts of fishing at a community level, and can be used to rank demersal fish communities across ocean regions for their comparative level of depletion, with important implications for conservation prioritization.

6.2 Future research directions

The bottom trawl survey dataset considered in this study was informative about changes to demersal fish communities, but there are other historical data available (chapter 2) that, if analyzed, may add to the picture of long-term changes in South East Australian marine ecosystems. These data include, for example, catch and effort data from shark-netting programs that have been in place since the 1940s along the New South Wales and Queensland coasts (Reid & Krogh, 1992). The full dataset has been recently digitalized and is now available for analysis, providing a basis for insights into long-term trends in shark species abundance. Also, semi-quantitative information that can be extracted from anecdotes and fisher interviews may be used to deepen our knowledge about past fishing practices, and the impacts they may have had on marine systems. For instance, the royal commission into the fisheries of Tasmania (1882) interviewed 13 citizens, from fishermen to Government statisticians and economic advisers, who provided details about fishing techniques, grounds and seasons, and about species caught and their quantities. Complete interviews are provided as part of the royal commission report, under the section 'minutes of evidence' (Fisheries Inquiry Commission, 1883), and summary tables (e.g. a table of the main species found in each fishing location) were created as part of this work, but not used in the present study due to time constraints. Repeating such interviews today with similar sets of stakeholders could shed light on the evolution of social and ecological aspects of marine systems.

International significance of the findings

Historical ecology has developed substantially since its first focus in the 1960s. The steep increase during the last 20 years in the number of papers that identify themselves as historical ecology is a clear sign of the growing importance of this relatively new field (Szabó, 2014, Fig. 1-1), which, through the mixing of multiple sources of information (e.g. long-term biodiversity datasets, archeological and genetic records), has pushed the chronological edges of our knowledge far back in time (e.g. Jackson & Johnson, 2001; Pandolfi *et al.,* 2003; Lotze *et al.,* 2006). Also, the interrelation of individual studies and findings in all parts of the world is growing (Szabó, 2014), including for the marine component of this field (Holm *et al.,* 2001, 2014; Máñez *et al.,* 2014). In marine science, global initiatives, such as the History of Marine Animal Populations (HMAP, 2001) and the Sea Around Us programs, have examined disparate socio-ecological systems (e.g. European Seas e.g. Leitão *et al.,* 2014; Piroddi *et al.,* 2015). Recently, a new global research network, the Ocean Past Initiative (OPI), has been proposed to assist in coordinating research efforts worldwide (Máñez *et al.,* 2014).

This thesis adds to the international historical ecology literature. First, it improved our understanding of the causes and rates of ecological changes in temperate fish communities of the Southern Hemisphere, which had been relatively little considered by historical ecologists. With some exceptions (e.g. Klaer, 2001; Carroll *et al.,* 2014), the quantitative component of historical ecology has focused on regions of the world where detailed long-term datasets are available because the fishing components have always been important, and thus better documented. For instance, the majority of studies providing estimates of rates of ecological change focus on marine systems of Europe (e.g. Ferretti *et al.,* 2008), and the North Atlantic (e.g. Lotze & Milewski, 2004; Alexander *et al.,* 2009). Second, because the South East Australian region provided patchy and unbalanced data that are more representative of available historical datasets in data poor regions, the approach used in this study could provide the impetus for the analysis of other analogous and overlooked historical datasets around the world.

Importantly, this study has shown that despite the limitations of historical data, an historical perspective and historical records are always worth exploring because they are able to shed light on important longer-term socio-ecological shifts, thus helping to redefine research and management priorities.

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8 Appendix 1 - to chapter 3

Table 8-1. Tows landmark positions in latitudinal and longitudinal decimal degrees (DD), positions at 200 m and 1000 m.

Landmark	Lat landmark	Long landmark	Lat 200m	Long 200m	Lat 1000m	Long 1000m
Babel Island	-39.95	148.33	-39.95	148.86	-39.95	148.96
Barrenjoey Point	-33.58	151.33	-33.58	151.9	-33.58	152.13
Bass Point	-34.6	150.08	-34.6	151.18	-34.6	151.33
Batemans Bay	-35.73	150.25	-35.73	150.83	-35.73	150.93
Bermagui	-36.43	150.06	-36.43	150.26	-36.43	150.33
Between Haystack Bay and North end of Twofold Bay	-39.93	148.01	-39.93	148.75	-39.93	148.95
Bird Island	-33.23	151.6	-33.23	152.23	-33.23	152.4
Botany Bay	-34	151.23	-34	151.55	-34	151.85
Broken Bay	-33.6	151.31	-33.6	151.86	-33.6	152.13
Broughton Island	-32.61	152.31	-32.61	152.66	-32.61	152.85
Broulee	-35.85	150.18	-35.85	150.51	-35.85	150.56
Brush Island	-35.53	150.41	-35.53	150.71	-35.53	150.78
Bulgo	-34.2	151	-34.2	151.4	-34.2	151.58
Cape Everard	-37.8	149.26	-38.11	149.36	-38.33	149.43
Cape Forster	-32.18	152.51	-32.18	152.9	-32.18	153.08
Cape Grenfell	-34.9	150.6	-34.9	151.05	-34.9	151.18
Cape Hawke	-32.21	152.56	-32.21	152.93	-32.21	153.08
Cape Howe	-37.5	149.98	-37.5	150.23	-37.5	150.3
Cape Three Points	-33.5	151.41	-33.5	151.93	-33.5	152.13
Catherine Hill Bay	-33.15	151.63	-33.15	152.28	-33.15	152.48
Charlott Head	-32.33	152.55	-32.33	152.9	-32.33	153.01
Coalcliff	-34.25	150.98	-34.25	151.4	-34.25	151.58
Coogee	-33.91	151.25	-33.91	151.56	-33.91	151.85
Cronulla	-34.03	151.18	-34.03	151.58	-34.03	151.86
Crookhaven River	-34.9	150.75	-34.9	151.08	-34.9	151.23

Crowdy Head	-31.85	152.75	-31.85	153.05	-31.85	153.31
Disaster Bay	-37.26	149.96	-37.26	150.28	-37.26	150.35
Eden	-37.06	149.91	-37.06	150.28	-37.06	150.35
Everard Light North East	-37.78	149.28	-37.78	150.11	-37.78	150.2
Flinders Island	-39.93	148.01	-39.93	148.75	-39.93	148.95
Gabo Island	-37.56	149.91	-37.56	150.21	-37.56	150.3
Goalen	-36.51	150.05	-36.51	150.33	-36.51	150.41
Green Cape	-37.21	150.03	-37.21	150.33	-37.21	150.38
Haystack Rock	-42.2	148.06	-42.2	148.53	-42.2	148.61
Jervis Bay	-35.11	150.76	-35.11	150.96	-35.11	151.1
Jervis Bay (within)	-35.03	150.44	-35.03	150.44	-35.03	150.44
Jibbon	-34.06	151.15	-34.06	151.48	-34.06	151.68
Kiama	-34.66	150.85	-34.66	151.16	-34.66	151.33
Korogoro Island	-31.05	153.03	-31.05	153.2	-31.05	153.31
Lakes Entrance	-37.86	148	-38.33	148.51	-38.48	148.73
Lily Vale	-34.2	151	-34.2	151.4	-34.2	151.61
Long Point	-33.75	151.25	-33.75	151.75	-33.75	152.08
Manning River	-31.86	152.7	-31.86	153.08	-31.86	153.25
Maria Island	-42.66	148.13	-42.66	148.4	-42.66	148.51
Marion Bay	-42.8	148	-42.8	148.35	-42.8	148.45
Marley Beach	-34.11	151.13	-34.11	151.45	-34.11	151.7
Merimbula	-36.9	149.93	-36.9	150.26	-36.9	150.33
Montague Island	-36.25	150.23	-36.25	150.3	-36.25	150.38
Montague Island North	-36.23	150.21	-36.23	150.35	-36.23	150.4
Montague Island South	-36.25	150.21	-36.25	150.33	-36.25	150.4
Moon Bay	-36.7	149.98	-36.7	150.26	-36.7	150.35
Moon Island	-33.08	151.66	-33.08	152.38	-33.08	152.56
Morna Point	-32.78	152.08	-32.78	152.65	-32.78	152.78
Moruya	-35.91	150.13	-35.91	150.46	-35.91	150.51
Mowarry Point	-37.15	150	-37.15	150.28	-37.15	150.35
N.E. flinders Island	-39.86	148.08	-39.86	148.71	-39.86	148.9
N.E. Broughton Island	-32.6	152.31	-32.6	152.8	-32.6	152.9
Narrabeen	-33.7	151.01	-33.7	151.76	-33.7	152.05
New Zealand Ground	-33.71	151.43	-33.63	151.86	-33.56	152.13

Newcastle	-32.93	151.76	-32.93	152.55	-32.93	152.65
Norah Head	-33.28	151.58	-33.28	152.18	-33.28	152.31
O'Hara Head	-35.56	150.38	-35.56	150.68	-35.56	150.75
Port Hacking	-34.06	151.1	-34.06	151.51	-34.06	151.81
Port Jackson	-33.81	151.28	-33.81	151.65	-33.81	151.93
Port Kembla	-34.48	150.91	-34.48	151.26	-34.48	151.43
Port Macquarie	-31.45	152.91	-31.45	153.13	-31.45	153.25
Port Stephens	-32.7	152.16	-32.7	152.68	-32.7	152.83
Red Head	-35.25	150.55	-35.25	150.9	-35.25	151
Seal Rock	-32.43	152.53	-32.43	152.81	-32.43	152.95
Shoalhaven Head	-34.85	150.75	-34.85	151.1	-34.85	151.21
Sisters	-39.5	147.73	-39.2	148.61	-39.13	148.73
Smoky Cape	-30.91	153.08	-30.91	153.2	-30.91	153.31
St Helens	-41.33	148.25	-41.33	148.61	-41.33	148.8
Storm Bay	-43.1	147.5	-43.5	147.81	-43.73	147.95
Sugar Rock	-33.11	151.55	-33.11	152.33	-33.11	152.55
Sydney Head	-33.85	151.3	-33.85	151.7	-33.85	152.08
Tathra Head	-36.73	149.98	-36.73	150.26	-36.73	150.33
The Pines	-36.01	150.15	-36.01	150.4	-36.01	150.46
Tollgate Island	-35.75	150.26	-35.75	150.56	-35.75	150.63
Tuggerah Lakes	-33.33	151.5	-33.33	151.11	-33.33	152.28
Tuncurry	-32.18	152.5	-32.18	152.93	-32.18	153.08
Twofold bay	-37.08	149.91	-37.08	150.28	-37.08	150.35
Ulladulla	-35.35	150.46	-35.35	150.81	-35.35	150.98
Wallis Lake	-32.26	152.51	-32.26	152.93	-32.26	153.06
Warden	-35.38	150.5	-35.38	150.78	-35.38	150.88
Wattamolla	-34.13	151.13	-34.13	151.46	-34.13	151.71
Wollongong	-34.41	150.9	-34.41	151.26	-34.41	151.46
Wreck Bay	-35.18	150.63	-35.18	150.91	-35.18	151.05
Wybung Head	-33.2	151.66	-33.2	152.23	-33.2	152.4

Table 8-2. Species names used in *historical survey*'s logs and corresponding old and current scientific names. Conversion from survey log name to old scientific names following *(a) Endeavour* biological records; *(b)* Stead, 1906; *(c)* Roughley *et al.,* 1916 and Roughley, 1953; *(d)* Farnell & Waite, 1898; and *(e)* Ogilby 1886.

Survey report name	Old scientific name	Scientific name
Amblyrhynchotus oblongus	*(a)Amblyrhynchotus oblongus*	*Tetraodontidae - undifferentiated*
Angel fish	*(d)Rhina squatina/squatina squatina*	*Squatina spp.*
Antennarius nummifer	*(a)Antennarius nummifer*	*Antennarius nummifer*
Anthias pulchellus	*(a)Anthias pulchellus*	*Lepidoperca pulchella*
Apogonops		*Apogonops spp.*
Apogonops anomalus	*(a)Apogonops anomalus*	*Apogonops anomalus*
Argentina	*(c)Monodactylus argenteus*	*Monodactylidae - undifferentiated*
Australian cod		*Moridae - undifferentiated*
Banded stingaree		*Urolophus cruciatus*
Barracouta	*(a)Thyrsites atun*	*Thyrsites atun*
Bass flathead	*(b)Platycephalus bassensis*	*Platycephalus bassensis*
Bastard trumpeter	*(b)Latris ciliaris*	*Latridopsis forsteri*
Beardie	*(b)Lotella callarias/rhacina*	*Lotella rhacina*
Bellow fish macrorhamphosus	*(a)Macrorhamphosus*	*Macroramphosus scolopax*
Bellows fish	*(a)Macrorhamphosus scolopax/gallinago*	*Macroramphosus scolopax*
Black sole	*(c)Synaptura nigra*	*Brachirus nigra*
Black stin ray	*(d)Trygon pastinaca*	*Dasyatidae - undifferentiated*
Boar fish	*(a)Zanclistius elevatus*	*Zanclistius elevatus*
Box fish	*(b)Ostraciontidae*	*Ostraciidae - undifferentiated*
Brachionichthys hirsutus	*(a)Brachionichthys hirsutus*	*Brachionichthys hirsutus*
Brown puller		*Chromis hypsilepis*
Bull's eye	*(b)Pempheridae spp..*	*Pempherididae - undifferentiated*
Bullrout	*(d)Centropogon robustus*	*Notesthes robusta*
C. ayraudi	*(b)Monacanthus ayraudi/chinaman*	*Nelusetta ayraud*
C. morwong	*(c)Cheliodactylidae*	*Cheilodactylidae - undifferentiated*
Callianthias platei	*(a)Callanthias platei*	*Callanthias spp.*
Carpet shark		*Order orectolobiformes - undifferentiated*

Centropercis nudivittis	*(a)Centropercis nudivittis*	*Champsodon spp.*
Cephaloscyllium		*Cephaloscyllium spp.*
Chimera		*Chimaeridae - undifferentiated*
Chinaman leatherjacket	*(b)Monacanthus ayraudi*	*Nelusetta ayraud*
Cod gurnard	*(c)Triglidae*	*Triglidae - undifferentiated*
Cod		*Moridae - undifferentiated*
Cod physiculus	*(a)Physiculus*	*Pseudophycis barbata*
Collared cat shark		*Scyliorhinidae - undifferentiated*
Common stingray		*Dasyatidae - undifferentiated*
Conger eel	*(a)Congrus habenatus*	*Congridae - undifferentiated*
Congermurena	*(a)Congermurena habenata*	*Congridae - undifferentiated*
Crested flounder	*(b)Lophonectes gallus*	*Lophonectes gallus*
Cristiceps argyropleura	*(a)Cristiceps argyropleura*	*Cristiceps argyropleura*
Cucumber fish	*(a)Chloropthalmus nigripinnis*	*Paraulopus nigripinnis*
Deepsea flathead	*(c)Neoplatycephalus macrodon*	*Platycephalus richardsoni*
Deepsea flute mouth	*(b)Fistularidae*	*Fistulariidae - undifferentiated*
Devil fish	*(e)Mobula mobular*	*Myliobatidae - undifferentiated*
Dog fish	*(a)Squalus megalops*	*Squalus spp.*
Dogfish squalus megalops	*(a)Squalus megalops*	*Squalus spp.*
Dragonet	*(b)Callionymidae*	*Draconettidae & callionymidae - undifferentiated*
Eagle ray	*(b)Myliobatis australis*	*Myliobatidae - undifferentiated*
Elephant fish		*Callorhinchus milii*
Emissola	*(e)Emissola ganearum/e.maugeana*	*Mustelus antarcticus*
Farnell's boar fish	*(a)Histiopterus farnelli*	*Paristiopterus labiosus*
Fiddler	*(b)Trygonorrhina fasciata*	*Trygonorrhina spp.*
Flathead	*(c)Platycephalidae*	*Platycephalidae - undifferentiated*
Flounder		*Pleuronectidae & others- undifferentiated*
Flounder multimaculatus	*(a)Pseunderhomhus multimaculatus*	*Pseudorhombus jenynsii*
Flounder pseunderhomhus multimaculatus	*(a)Pseunderhomhus multimaculatus*	*Pseudorhombus jenynsii*
Flying gurnard	*(d)Trigla polyommata*	*Pterygotrigla polyommata*
Fortescue	*(d)Pentaroge marmorata*	*Gymnapistes marmoratus*
Ghost fish		*Hydrolagus ogilbyi*
Grey banded perch	*(b)Serranidae family*	*Serranidae - undifferentiated*

Grey nurse shark	*(d)Odontaspp.s americanus*	*Odontaspididae - undifferentiated*
Gummy		*Mustelus spp.*
Gurnard kumu	*(a)Kumu- chelidonichthys kumu*	*Chelidonichthys kumu*
Gurnard polyommata	*(a)Polyommata- pterygotrigla polyommata*	*Pterygotrigla polyommata*
Hake		*Merlucciidae & macruronidae - undifferentiated*
Hake jordanidia		*Merlucciidae & macruronidae - undifferentiated*
Hammerheaded shark	*(d)Zygaena malleus*	*Sphyrna zygaena*
Horse mackerel	*(a)Trachurus declivis*	*Trachurus declivis*
Jackass fish	*(c)Dactylospp.rus macropterus*	*Nemadactylus macropterus*
Javelin fish	*(a)Chilomycterus jaculiferus*	*Diodontidae*
John silver dory		*Zeidae & cyttidae - undifferentiated*
John dory	*(a)Zeus faber*	*Zeus faber*
Keel headed parrot fish	*(b)Labridae & scaridae*	*Labridae - undifferentiated*
Knight fish	*(b)Monocentris gloria-maris*	*Monocentrididae - undifferentiated*
Kumu	*(c)Chelidonichthys kumu*	*Chelidonichthys kumu*
Lagocephalus lunaris	*(a)Lagocephalus lunaris*	*Lagocephalus lunaris*
Large toothed flounder	*(b)Paralichthys arsius*	*Pseudorhombus arsius*
Latchet		*Pterygotrigla polyommata*
Lead coloured dory		*Zeidae & cyttidae - undifferentiated*
Leather jackets	*(b)Monacanthus ayraudi*	*Nelusetta ayraud*
Ling	*(b)Lotella callarias/rhacina*	*Lotella rhacina*
Lizard fish		*Bathysauridae & synodontidae - undifferentiated*
Long nosed flounder	*(c)Ammotretis rostratus*	*Ammotretis rostratus*
Lophonectes	*(a)Lophonectes*	*Lophonectes gallus*
Mackerel	*(d)Scomber antarcticus*	*Scomber australasicus*
Monochanthu mosaicus	*(a)Monochanthu mosaicus*	*Eubalichthys mosaicus*
Monochanthu setosus	*(a)Monochanthu setosus*	*Meuschenia scaber*
Morwong	*(a)Dactylospp.rus carponemus*	*Nemadactylus douglasi*
Mustelus		*Mustelus spp.*
Nannygai	*(d)Beryx affinis*	*Centroberyx affinis*
Nany banded sole		*Soleidae - undifferentiated*
Narrow banded sole	*(b)Solea macleayana*	*Synclidopus macleayanus*
Nemadactylus morwong	*(a)Nemadactylus morwong*	*Nemadactylus macropterus & nemadactylus spp.*
Numb fish		*Narcinidae - undifferentiated*

Old wife	*(d)Enoplosus armatus*	*Enoplosus armatus*
Orange perch	*(e) Anthias pulchellus*	*Lepidoperca pulchella*
Orectolobus	*(a)Orectolobus*	*Orectolobidae - undifferentiated*
Other gurnard	*(c)Triglidae*	*Triglidae - undifferentiated*
Paralichthys tenuirastrum	*(a)Paralichthys tenuirastrum*	*Pseudorhombus tenuirastrum*
Parapercis	*(a)Parapercis*	*Parapercis spp.*
Parapercis allporti	*(a)Callanthias allporti*	*Callanthias spp.*
Parapercis ocularis	*(a)Parapercis ocularis*	*Parapercis spp.*
Paratrachichthys trailli	*(a)Paratrachichthys trailli*	*Paratrachichthys macleayi*
Parrot fish	*(a)Pseudolabrus spp.*	*Labridae - undifferentiated*
Perch		*Serranidae - undifferentiated*
Pike	*(b)Sphyraena novaehollandiae*	*Sphyraena novaehollandiae & dinolestes lewini*
Pilchard		*Sardinops sagax*
Polyoammata	*(c)Pterygotrigla polyoammata*	*Pterygotrigla polyommata*
Porcupine	*(b)Dicotylichthys punctulatus*	*Diodontidae - undifferentiated*
Port jackson shark	*(b)Heteroontus philippi*	*Heterodontus spp.*
Rays		*Dasyatidae - undifferentiated*
Red bull s eye	*(b)Pempheris spp..*	*Pempheris spp.*
Red cod		*Scorpaena papillosa*
Red gurnard	*(c)Triglidae*	*Triglidae - undifferentiated*
Red gurnard perch	*(c)Triglidae*	*Triglidae - undifferentiated*
Red gurnet perch	*(c)Helicolenus percoides*	*Helicolenus percoides*
Red morwong	*(b)Cheilodactylus fuscus*	*Cheilodactylus fuscus*
Red mullet	*(b)Upeneus porosus*	*Upenichthyes spp.*
Red perch	*(a)Caesioperca rasor*	*Caesioperca rasor*
Red rock cod	*(d)Scorpaena cruenta*	*Scorpaena spp.*
Rock cod	*(a)Pseudophycis barbata/physiculus barbatus*	*Pseudophycis barbata*
Rock cod tasmanian		*Scorpaenidae - undifferentiated*
Rock perch and allports perch	*(a)Callanthias allporti*	*Callanthias spp.*
Rough billied pipe fish		*Syngnathidae - undifferentiated*
Sand flathead	*(c)Platycephalidae*	*Platycephalidae - undifferentiated*
Sand whiting	*(c)Sillago ciliata*	*Sillago ciliata*
Sargeant baker	*(a)Aulopus purpurissatus*	*Aulopus purpurissatus*

Saury		*Saurida spp.*
Saw shark		*Pristiophoridae - undifferentiated*
Sawfish	*(b)Pristis zysron*	*Pristiophorus spp.*
Schnapper	*(b)Pagrosomus auratus*	*Pagrus auratus*
School shark	*(d)Galeus australis*	*Galeorhinus galeus*
Sea perch	*(a)Helicolenus percoides*	*Helicolenus percoides*
Sea pike	*(a)Sphyraena*	*Sphyraena spp.*
Sergeant baker	*(a)Aulopus purpurissatus*	*Aulopus purpurissatus*
Shark		*Class chondrichthyes*
Shovel nose ray	*(d)Rhinobatus granulatus*	*Rhinobatidae - undifferentiated*
Silver belly	*(b)Gerridae*	*Gerreidae - undifferentiated*
Silver belly victoria	*(b)Gerridae*	*Parequula melbournensis*
Silver bream	*(e)Gerres ovatus*	*Gerres subfasciatus*
Silver dory	*(a)Cyttus australis*	*Cyttus australis*
Silversides	*(c)Gerridae*	*Atherinidae - undifferentiated*
Skate		*Rajidae - undifferentiated*
Skipjack	*(c)Pomatomus saltatrix*	*Pseudocaranx spp.*
Small shark		*Class chondrichthyes*
Small toothed flounder	*(a)Pseudorhombus multimaculatus*	*Pseudorhombus jenynsii*
Snapper	*(c)Pagrosomus auratus*	*Pagrus auratus*
Sole		*Soleidae - undifferentiated*
Spikies		*Squalus spp.*
Spiny sea horse		*Solegnathus spp.*
Spotted cat shark		*Scyliorhinidae - undifferentiated*
Spotted flounder	*(a)Paralichthys novaecambriae*	*Pleuronectidae - undifferentiated*
Spriny dog		*Squalus spp.*
Star grazer	*(b)Anema inerme*	*Uranoscopidae - undifferentiated*
Stingrays	*(d)Trygon pastinaca*	*Dasyatidae - undifferentiated*
Stonelifter	*(b)Kathetostoma laeve*	*Uranoscopidae - undifferentiated*
Synodus tumbil	*(a)Synodus tumbil*	*Synodontidae - undifferentiated*
Tailor	*(d)Temnodon saltator*	*Pomatomus saltatrix*
Tasmanian black perch jackass	*(a)Chilodactylus macropterus*	*Nemadactylus macropterus*
Tasmanian flounder		*Rhombosolea tapirina*
Tasmanian numb fish		*Narcine tasmaniensis*

Tasmanian red perch		*Neosebastes spp.*
Tasmanian silver belly	*(b)Gerridae*	*Parequula melbournensis*
Teraglin	*(d)Otolithus atelodus*	*Atractoscion aequidens*
Thetis fish	*(a)Sebastes thetidis*	*Neosebastes thetidis*
Tigers	*(c)Neoplatycephalus macrodon*	*Platycephalus richardsoni*
Tigers flatehead	*(c)Neoplatycephalus macrodon*	*Platycephalus richardsoni*
Trachinocephalus myops	*(a)Trachinocephalus myops*	*Trachinocephalus myops*
Trevalla	*(a)Seriolella brama/punctata*	*Seriolella brama & seriolella punctata*
Trevally	*(a)Caranx spp.*	*Caranx georgianus*
Trumpeter	*(a)Latris spp.*	*Latridae - undifferentiated*
Trumpeter tasmania	*(b)Latris hecateia*	*Latris lineata*
Trumpeter bastard	*(b)Latris ciliaris*	*Latridopsis forsteri*
Trumpeter perch	*(c)Pelates sexlineatus*	*Pelates sexlineatus*
Trumpeter whiting	*(d)Sillago bassensis*	*Sillago bassensis*
Whiptail	*(b)Macruridae*	*Coelorinchus spp.*
Whiting	*(a)Sillago bassensis*	*Sillago bassensis*
Whiting grass	*(e) sillago ciliata*	*Sillago spp.*
Whiting sea	*(a)Sillago bassensis*	*Sillago bassensis*
Wirrah	*(c)Acanthistius serratus*	*Acanthistius serratus*
Wobbegong	*(d)Crossorhinus barbatus*	*Orectolobidae - undifferentiated*
Yellowtail	*(d)Trachurus declivis*	*Trachurus spp.*
Zanclutius		*Zanclistius spp.*

9 Appendix 2 - to chapter 4

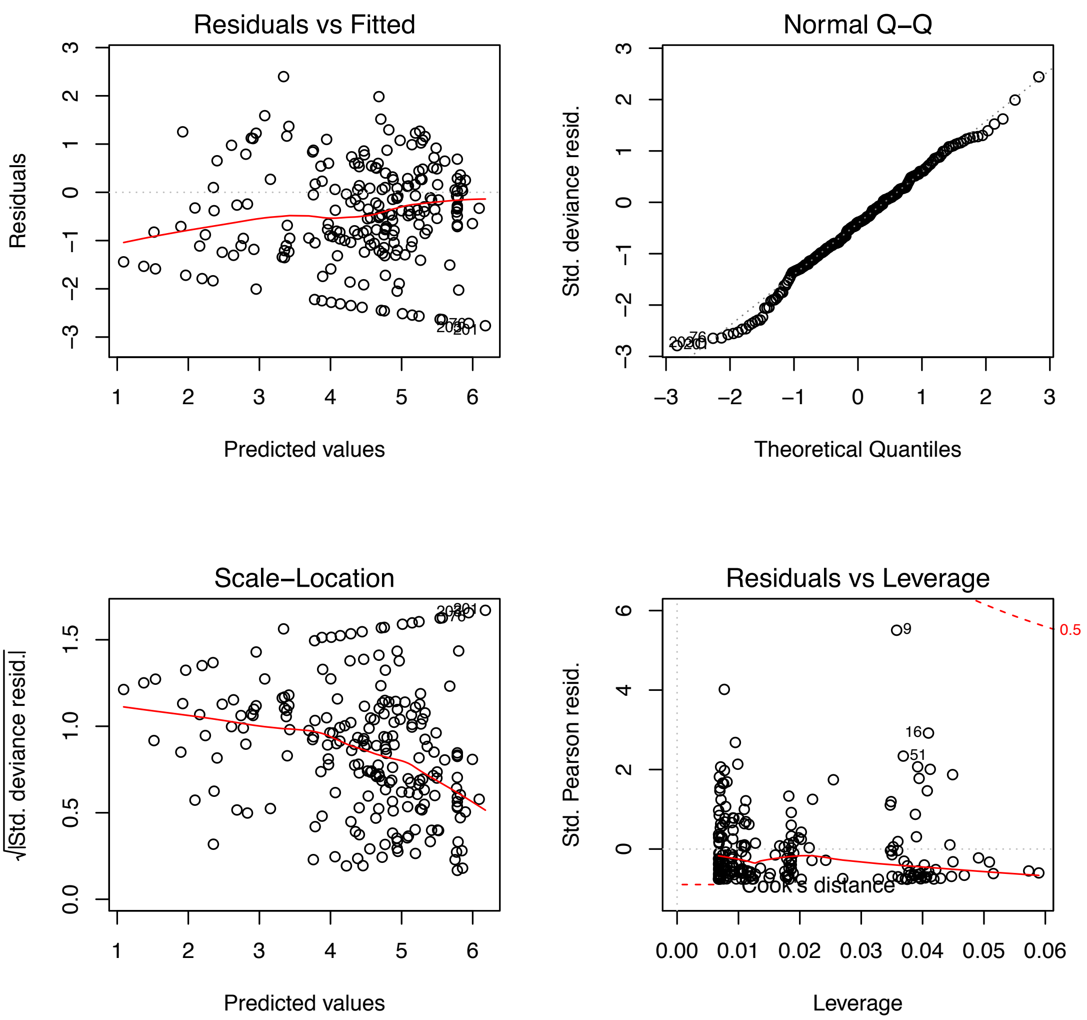

Figure 9-1. GLM diagnostic plots for the gurnard family.

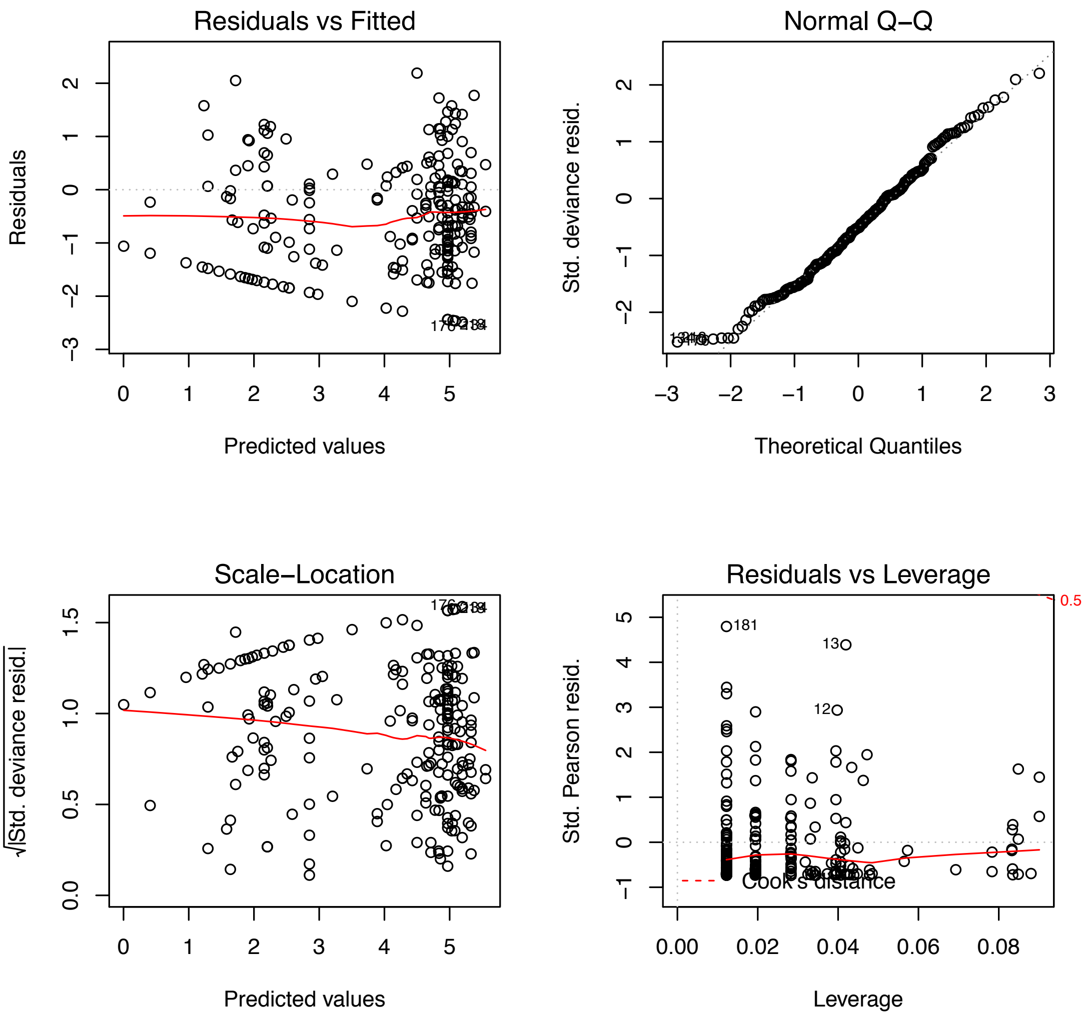

Figure 9-2. GLM diagnostic plots for the flathead family.

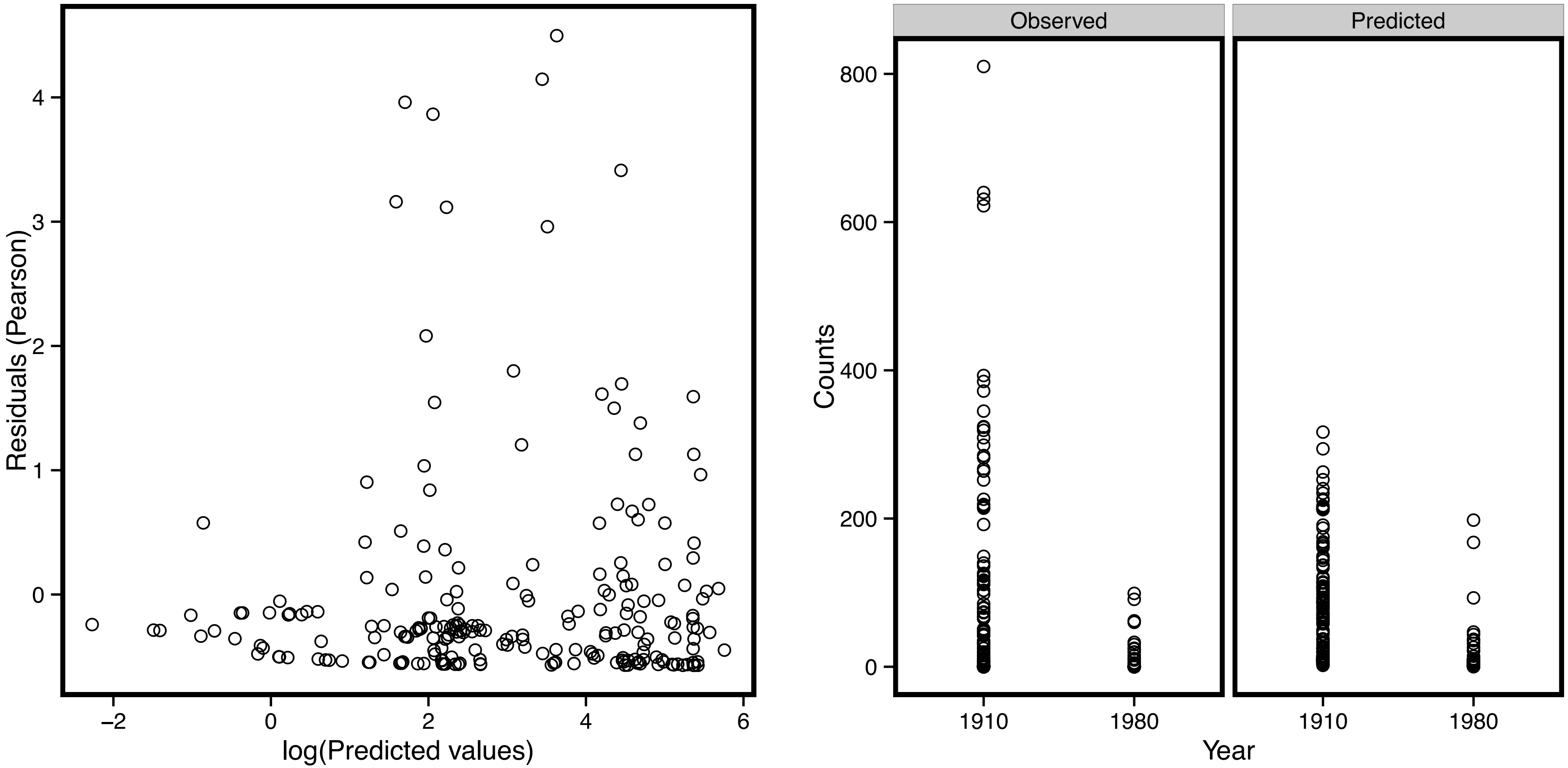

Figure 9-3. GLM diagnostic plots for the morwong family.

10 Appendix 3 – to chapter 5

10.1 Surveys datasets and data standardization

The first bottom trawl survey in Australian waters was carried out on the continental shelf off Sydney, in 1898 (Farnell & Waite, 1898), and since then, surveys have been performed intermittently along the coast of South East Australia. However taxonomic resolution and whether all or a fraction of the sampled species were reported, changed over the years. Hence I decided to focus our analysis on the period between 1976 and 1997, when the species identification was consistently at the species level and all the sampled species were reported.

I used trawl survey data from three research agencies:

1) New South Wales Department of Primary Industries (NSW DPI). The NSW DPI database includes a collection of surveys carried out between 1976 and 1997, with the *F.R.V Kapala* (e.g. Graham *et al.,* 1997, 2001). The vessel surveyed the continental shelve and slope along the New South Wales coast (Fig. 5-2). As survey objectives changed over time, I selected a subset of surveys with the specific aim to explore demersal fish communities. Seven surveys were selected. Each of these surveys was conducted over two years and used a stratified sampling design. Stratification was by depth and geographical location.

Specifically, the survey "*Kapala* 1993-95" covered the continental shelf of New South Wales (depths shallower than 200 m) and explored three inshore (30-60 m), three mid-shelf (100-125 m) and three outer-shelf (130-150 m) grounds. The surveys "*Kapala* 1976-77", "*Kapala* 1978-80" and "*Kapala* 1996-97" (Andrew *et al.,* 1997) covered the continental upper slope (200-650 m) and explored grounds off Sydney

(North region), Ulladulla (Central region) and Eden (South region). The surveys "*Kapala* 1983-84" and "*Kapala* 1986-87" covered the continental mid-slope (depths greater than 650 m) and explored grounds off Shoalhaven (North region) and Ulladulla (Central region). The survey "*Kapala* 1978-79" covered both continental shelf and upper slope and was confined to the northern region of New South Wales.

To obtain homogeneous spatio-temporal strata covering the coast of New South Wales I divided the dataset according to: 1) survey; 2) depth stratum reported in each survey (e.g. inshore shelf, mid-shelf and outer shelf); and 3) coastal regions (North, Central and South coast of New South Wales). As my aim was to obtain spatio-temporal strata with enough samples to build complete SAC, I exclude strata with less than 20 tows. I obtained 23 strata (Table 5-1).

2) Institute for Marine and Antarctic Studies (IMAS). The IMAS database includes a collection of bottom trawl surveys carried out by the Division of Sea Fisheries (formerly Tasmanian Fisheries Development Authority and later Department of Sea Fisheries). A combination of research and fishing vessels (e.g. F.R.V. *Challenger* and the commercial vessel *Mary Belle*), and nets were used. Surveys were carried out between 1975 and 1995, with the aim of exploring the demersal resources of the continental shelf and upper slope of Tasmania (Fig. 5-2).

Due to poor species list and overall low data quality, I excluded two surveys from all of my analysis: the oldest one, carried out in 1975-76 with F.R.V. *Zeehaan*; and an inshore survey, carried out in 1980 with R.V *Mary Belle*. Data between 1979 and 1987, summarized in Lyle (1993), consisted of tows randomly allocated along the coast of Tasmania. Some of these tows were part of planned surveys, others were carried out as groups of "exploratory tows". No bottom trawl surveys were carried out between 1983 and 1993. In 1993 the *F.R.V. Challenger* extensively surveyed the south and eastern section of the continental shelf of Tasmania (Jordan, 1997).

To obtain homogeneous spatio-temporal strata covering the coast of Tasmania I divided the dataset according to 1) groups of 3 to 4 years (1979-1982; 1983-1985 and 1993-1995); 2) depth strata (continental shelf – shallower than 200 m, and continental slope – deeper than 200 m); and 3) coastal regions (North, South, West and East coasts of Tasmania). Due to low tow density, I could not stratify the IMAS dataset using the same stratification by depth method applied to NSW DPI dataset, which included multiple depth strata for the continental shelf and slope.

The IMAS database contained tows for which location was mistaken or incorrectly recorded. Specifically, tows' reported latitude and longitude placed them on the continental shelf, whereas tows' reported depth located them on the continental slope. I excluded from my analysis strata in which more than 1/3 of the tows' location was mistaken. As for the NSW DPI dataset, I excluded strata with less than 20 tows. I obtained 8 spatio-temporal strata (Table 5-1).

3) Commonwealth Scientific and Industrial Research Organization (CSIRO). The CSIRO database includes a collection of surveys carried out during the 80s, with the *F.R.V. Soela* (e.g. Koslow *et al.,* 1994). These surveys mainly focused on orange roughy's (*Hoplostethus atlanticus*) distribution and biology, yet all species caught during sampling were reported. *Soela* surveys covered the continental slopes around Tasmania, although few tows (22) were carried out off Ulladulla, in New South Wales (Fig. 5-2).

To obtain homogeneous spatio-temporal strata covering the coast of Tasmania and

New South Wales I divided the dataset according to 1) year/s (1984 and 1988-1989); 2) depth strata (continental shelf and continental slope); and 3) coastal regions (North, South, West and East coast of Tasmania, and central coast of New South Wales). As for the IMAS dataset, I could not stratify the CSIRO dataset using the same stratification by depth applied to NSW DPI dataset. I obtained 4 spatio-temporal strata (Table 5-1).

Additional strata sampling specifics, such as mean tow duration and nets characteristics, can be found in table 10-2.

For all surveys I had to crosscheck for species names and identification codes no longer in use. I updated species names and codes following the Codes for Australian Aquatic Biota (CAAB) (Yearsley *et al.,* 1997).

I could not control for all taxonomic bias across surveys. Taxonomic resolution was different across datasets. A total of 1984 records over 53371 (3.7%) were reported at higher taxonomic level than species. These records were reported as mix of families, family or genus (e.g. *Chlorophthalmidae & Paraulopidae & Bathysauroididae & Bathysauropsidae; Melanostomiidae; Raja spp.*). As I considered such broad classifications inadequate for my analysis, I delete these records from the datasets. A total of 1.8%, 4.9% and 18.3% of the records were excluded from NSW DPI, IMAS and CSIRO datasets, respectively. Aware that the high number of records deleted from CSIRO dataset may have impacted my results, I repeated my analysis but excluded data from CSIRO dataset. Results were consistent.

Species identification skills and taxonomic resolution may have also increased with time leading to higher number of species recorded in more recent periods. As an example, Endeavour dogfish (*Centrophorus harrissoni*) and Southern Dogfish (*Centrophorus uyato*) were reported as a single species in 1976 and acknowledged as two different species afterwards (Graham *et al.,* 2001). I have accounted for this difference in species classification by combining the two species. However, I could not account for unreported differences in species classification and for gradual taxonomic improvements over time, and thus could not entirely eliminate taxonomic bias.

Swept area per tow is a fundamental piece of information required to calculate SAC. When precise net and effort specifications were missing from survey datasets, I looked for essential information to estimate tow swept area (e.g. net opening and time trawled) in surveys' report.

I estimate swept area following Sparre & Venema (1989):

$$A = D * hr * X2 \tag{10-1}$$

$$D = V * t \tag{10-2}$$

Where V is the trawling speed; hr is the length of the head-rope; t is the time spent trawling. $X2$ is that fraction of the headrope length, hr, which is equal to the width of the path swept by the trawl, the 'wing spread', and its suggested value is 0.5.

10.2 Species-specific index of aggregation

To estimate a species-specific indices of aggregation I fitted a generalized linear model (GLM) with NBD to the catch data (number of individuals) of species sampled at least ten times. Hence, I assumed that the probability of obtaining n individuals of a species in any tow_i (fishing operation) followed a NBD, with mean μ_i and variance μ_{i+}

μ_i^2 k. The dispersion parameter of the distribution (k) reflects the degree of clustering between individuals (Pielou, 1977; Taylor *et al.,* 1979) and is estimated from the data. I modeled log(μ_i) as:

$$\log(\mu_i) = \alpha + \beta * X + \log\,(A_i) \tag{10-3}$$

where α is the intercept, X the matrix of covariates, β the vector of their relative parameters, and A_i is the swept area, treated as an offset. The matrix of covariates, X, included the tow latitude and depth. I did not include net characteristics because they remain constant within each of these strata. I considered main effects, the interaction between the two covariates and quadratic functions of both covariates. Quadratic functions were included to capture the tendency of animals to aggregate around optimal values of environmental variables. For practicality, I applied the same model to a large number species (i.e. 39 for strata 17 and 34 for strata 23, Fig. 10-2). Hence, model misspecification may have affected the k estimates, which needed to be treated as a rough index of intra-specific aggregation (Taylor *et al.,* 1979). However, as I was interested in detecting changes in aggregation across strata, a rough index of aggregation would nonetheless fit my purpose. To estimate an index of aggregation for each stratum (k_w) I averaged the species-specific k (see Figs.10-2 and 10-3 for species-specific k and stratum-specific k_w estimates, and model diagnostic for two representative species, respectively.)

10.3 SAC function selection

SAC is commonly described by a power (eqn. 5-1) or exponential (eqn. 5-2) function. These functions represent the most popular of many other models that have been discussed in the literature (Flather, 1996; Tjørve, 2003). To test whether eqn. 5-1 or eqn. 5-2 were the functions best describing SACs calculated from trawl survey data, I tested the fit of these and a series of other functions used to describe SAC (Flather, 1996; Tjørve, 2003). Functions main specifications are reported in Table 10-3, but see Tjørve (2003) for further details of all models. I then used model selection to identify the model best describing my SACs. Model selection and process uncertainty estimation were based on a multi-model information theoretic approach (Burnham & Anderson, 2002). For each SAC model m$_i$ I calculated the Akaike Information Criterion (AIC), AIC differences (Δ_i = AIC$_i$ – AIC$_{min}$) and Akaike weights:

$$w_i = \frac{\exp\,(-1/2\,\Delta\, i)}{\sum_{r=1}^{R} \exp\,(-1/2\,\Delta\, i)} \qquad (10\text{-}4)$$

Where *R* is the number of models fitted. Akaike weights are interpreted as probabilities of a given model being the best in explaining the data within a predefined set of alternative models. For each stratum, I selected the best model corresponding to the maximum Akaike weights.

Model selection analyses confirmed that eqn. 5-2 was the function that best described SACs calculated from trawl survey data. For 27 strata out of 35, eqn. 5-2 showed Akaike weights equal to one, indicating that this function had a 100% chance of being the function best fitting the data among those considered. For other two strata eqn. 5-2 showed Akaike weights equal to 0.82 and 0.64. For the remaining six strata, other functions were preferred. In particular, eqn. 5-1, rational and monod functions were the preferred function in 3, 2 and 1 of the strata, respectively.

Table 10-1. Parameter values used to simulate 40 communities.

Community	Richness (R)	Abundance (N)	Evenness (J')	Evenness (k_b)	Aggregation (k_w)
1	84	22798	0.71	0.5	0.6
2	85	22798	0.71	0.5	0.6
3	87	22798	0.71	0.5	0.6
4	88	22798	0.71	0.5	0.6
5	90	22798	0.71	0.5	0.6
6	91	22798	0.71	0.5	0.6
7	93	22798	0.71	0.5	0.6
8	94	22798	0.71	0.5	0.6
9	96	22798	0.71	0.5	0.6
10	97	22798	0.71	0.5	0.6
11	84	22798	0.71	0.5	0.6
12	84	26772	0.71	0.5	0.6
13	84	30747	0.71	0.5	0.6
14	84	34721	0.71	0.5	0.6
15	84	38696	0.71	0.5	0.6
16	84	42670	0.71	0.5	0.6
17	84	46645	0.71	0.5	0.6
18	84	50619	0.71	0.5	0.6
19	84	54594	0.71	0.5	0.6
20	84	58568	0.71	0.5	0.6
21	84	22798	0.58	0.3	0.6
22	84	22798	0.59	0.32	0.6
23	84	22798	0.61	0.34	0.6
24	84	22798	0.64	0.37	0.6
25	84	22798	0.65	0.39	0.6
26	84	22798	0.66	0.41	0.6
27	84	22798	0.67	0.43	0.6
28	84	22798	0.69	0.46	0.6
29	84	22798	0.7	0.48	0.6
30	84	22798	0.72	0.5	0.6
31	84	22798	0.71	0.5	0.01
32	84	22798	0.71	0.5	0.18
33	84	22798	0.71	0.5	0.34
34	84	22798	0.71	0.5	0.51
35	84	22798	0.71	0.5	0.67
36	84	22798	0.71	0.5	0.84
37	84	22798	0.71	0.5	1
38	84	22798	0.71	0.5	1.17
39	84	22798	0.71	0.5	1.33
40	84	22798	0.71	0.5	1.5

Table 10-2. Strata sampling specifics. SD indicates Standard Deviation.

Strata	N. of tows	Mean tow duration (h) ± SD	Mean tow speed (km/h) ± SD	*Head-rope* lengths (m)	*Cod-end* mesh sizes (mm)
1	47	1.1±0.3	5.6	21	90
2	44	2±0.2	4.4±0.5	21 & 56	90 & 42
3	43	2±0.2	4.3±0.4	21 & 56	90 & 42
4	68	1.7±0.5	4.3±0.4	30	90 & 42
5	95	1.7±0.5	4.3±0.4	30	90 & 42
6	64	1	5.6	56	42
7	67	1	5.6	56	42
8	69	1	5.6	56	42
9	69	1	5.6	56	42
10	82	1	5.6	56	42
11	74	1	5.6	56	42
12	64	1	5.6	56	42
13	67	1	5.6	56	42
14	64	1	5.6	56	42
15	88	1.1±0.3	5.6	21	90
16	82	1.2±0.4	5.6	21	90
17	63	1±0.2	5.6	21	90
18	72	1.3±0.4	5.6	21 & 56	90
19	45	1.4±0.5	5.6	21 & 56	90
20	80	1.1±0.2	5.6	21 & 56	90
21	54	1±0.1	5.6	21	90
22	48	1	5.6	21	90
23	63	1±0.1	5.6	21	90
24	97	1.8±0.8	5.6	Multiple from 24 to 60	90 & 110
25	20	1.3±0.8	5.6	Multiple from 22 to 41	90
26	52	1.9±0.7	5.6	Multiple from 22 to 38	90
27	51	2.6±1.2	5.6	Multiple from 22 to 60	90
28	25	1.5±0.4	5.6	Multiple from 34 to 56	90
29	45	2.4±1	5.6	Multiple from 53 to 60	90
30	112	0.5	5.4±0.3	26	20
31	125	0.5	5.5±0.2	26	20
32	22	1.4±0.9	4.4±1.3	35	40
33	42	0.9±0.3	4.3±2	26 & 35	40
34	54	0.8±0.1	4.3±1.5	35	40
35	105	0.8±0.1	4.7±1.5	35	40

Table 10-3. Functions commonly fit to SAC and tested in this study.

Name	Formula	Parameters	Shape
Power	$S = c*A^z$	2	Convex
Exponential	S = c + z*log(A)	2	Convex
Negative exponential	S = c(1 – exp(-z*A))	2	Convex
Monod	S = c*A/(z + A)	2	Convex
Rational function	S = (c + z*A)/(1 + f*A)	3	Sigmoid

Table 10-4. Richness, abundance, evenness and intra-specific aggregation for strata surveyed by the New South Wales Department of Primary Industry (NSW DPI). For intra-specific aggregation I reported Standard Errors (SE) and the number of species-specific *k* calculated (N. Species).

Strata	Richness (R)	Abundance (N)	Evenness (J')	Aggregation (kw) ± SE ± (N. Species)
1	152	25701	0.56	0.8 ± 0.07 (28)
2	124	16696	0.71	0.99 ± 0.06 (40)
3	121	14233	0.71	0.95 ± 0.06 (36)
4	134	30070	0.68	0.87 ± 0.06 (52)
5	161	37752	0.66	0.73 ± 0.05 (63)
6	131	356523	0.4	0.5 ± 0.05 (57)
7	80	391299	0.47	0.79 ± 0.09 (36)
8	94	465004	0.32	0.62 ± 0.08 (41)
9	101	415392	0.42	0.67 ± 0.08 (42)
10	129	546124	0.39	0.85 ± 0.08 (41)
11	119	75118	0.49	0.53 ± 0.07 (41)
12	97	357088	0.44	0.69 ± 0.07 (48)
13	87	390928	0.48	0.64 ± 0.07 (42)
14	112	191455	0.43	0.72 ± 0.07 (46)
15	114	71177	0.45	0.66 ± 0.06 (45)
16	115	50803	0.62	0.7 ± 0.06 (50)
17	84	22798	0.71	0.67 ± 0.05 (39)
18	107	91118	0.34	0.48 ± 0.07 (30)
19	89	42765	0.57	0.73 ± 0.07 (31)
20	89	45690	0.52	0.61 ± 0.07 (35)
21	106	39991	0.58	1.06 ± 0.07 (30)
22	107	31363	0.63	1 ± 0.08 (30)
23	97	58568	0.55	0.87 ± 0.08 (34)

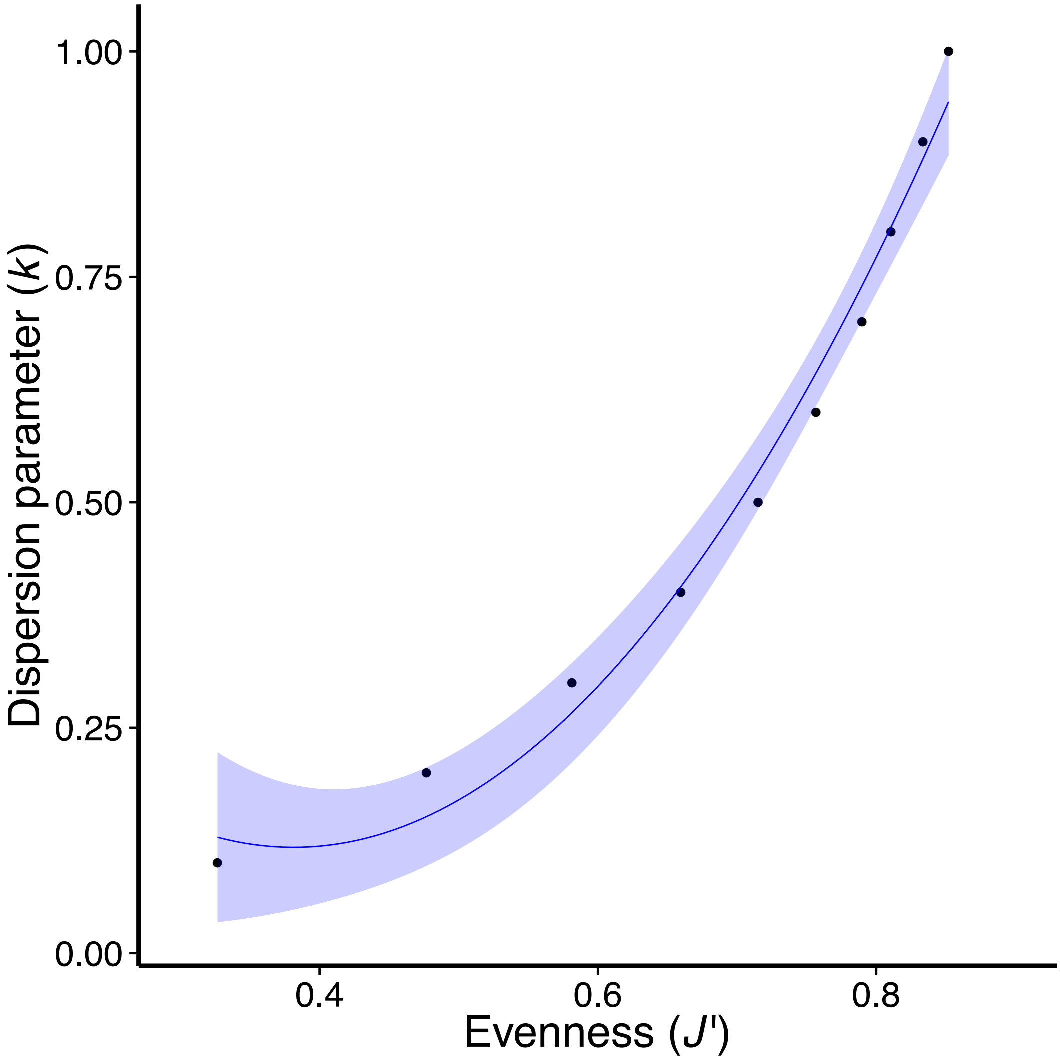

Figure 10-1. Relationship between the dispersion parameter of a zero truncated negative binomial distribution (TNBD) used to model community evenness (k_b), and Pielou's even index *J'* calculated for the generated community.

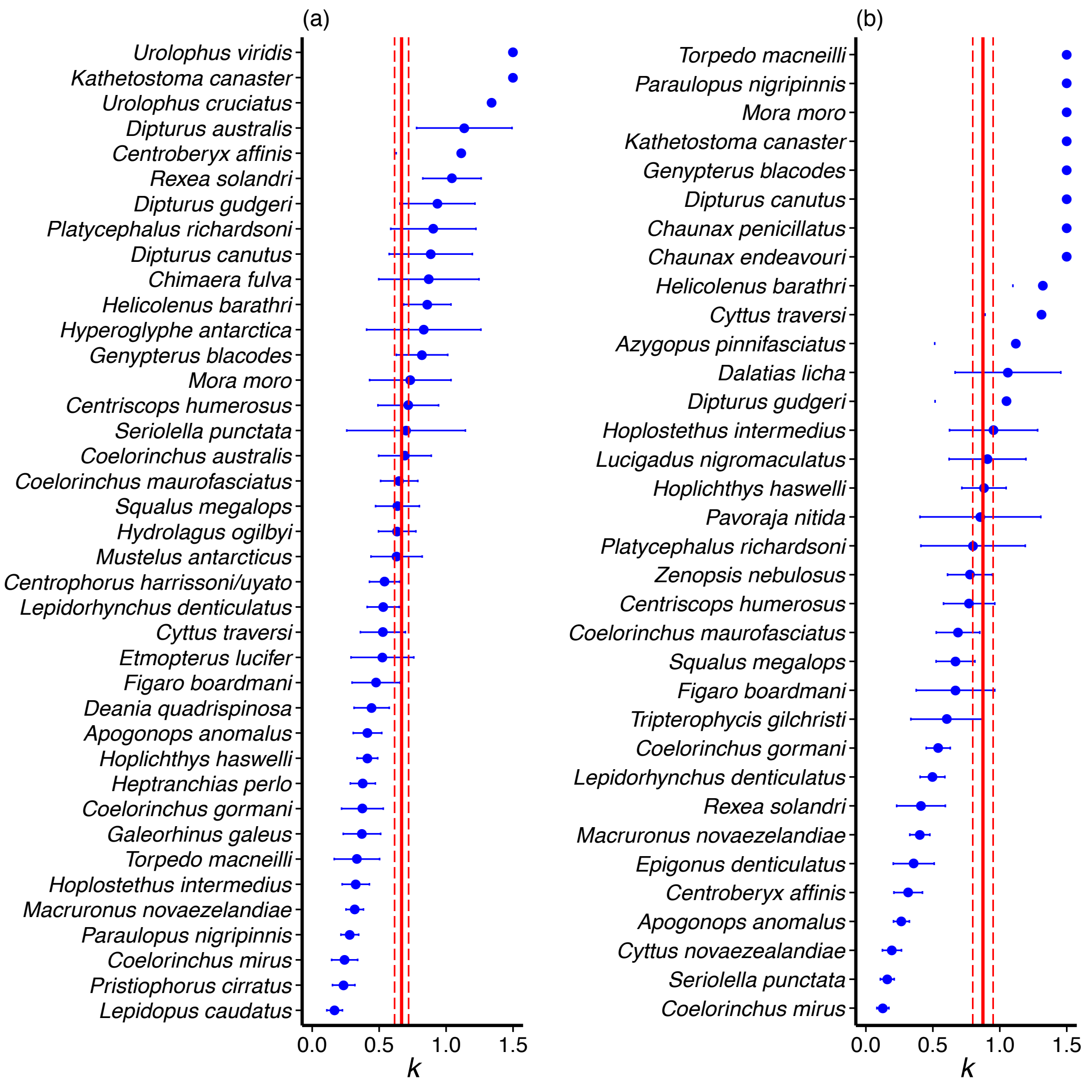

Figure 10-2. Species-specific values of k (blue dots) and stratum-specific values of k_w (vertical line) for strata (a) 17 and (b) 23. Standard errors for k and k_w are shown.

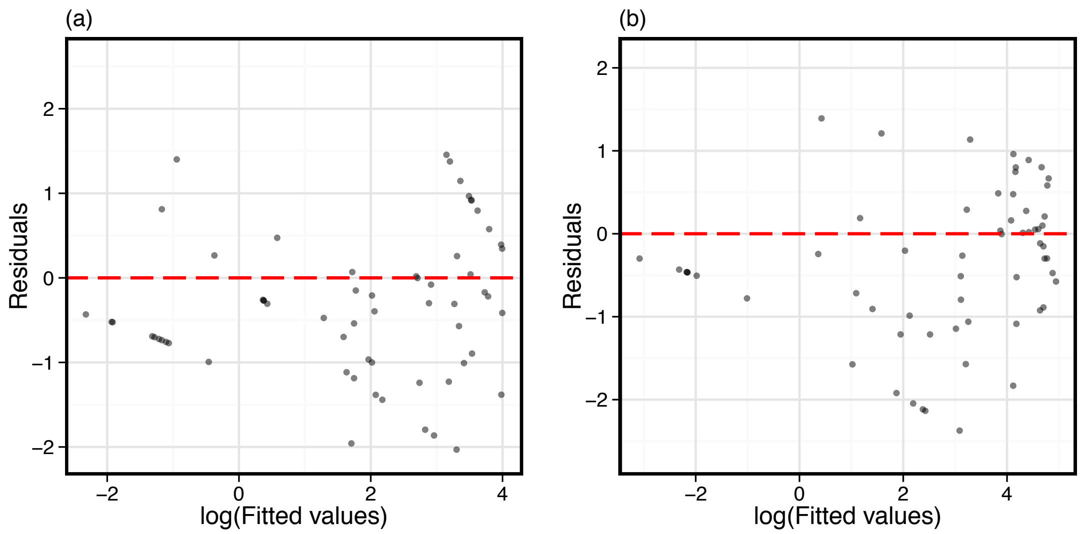

Figure 10-3. Generalized linear model (GLM) residuals versus fitted values for two of the most abundant species sampled in stratum 17. (a) Silver gemfish (R*exea solandri*), and (b) bigeye sea perch (*Helicolenus barathri*).

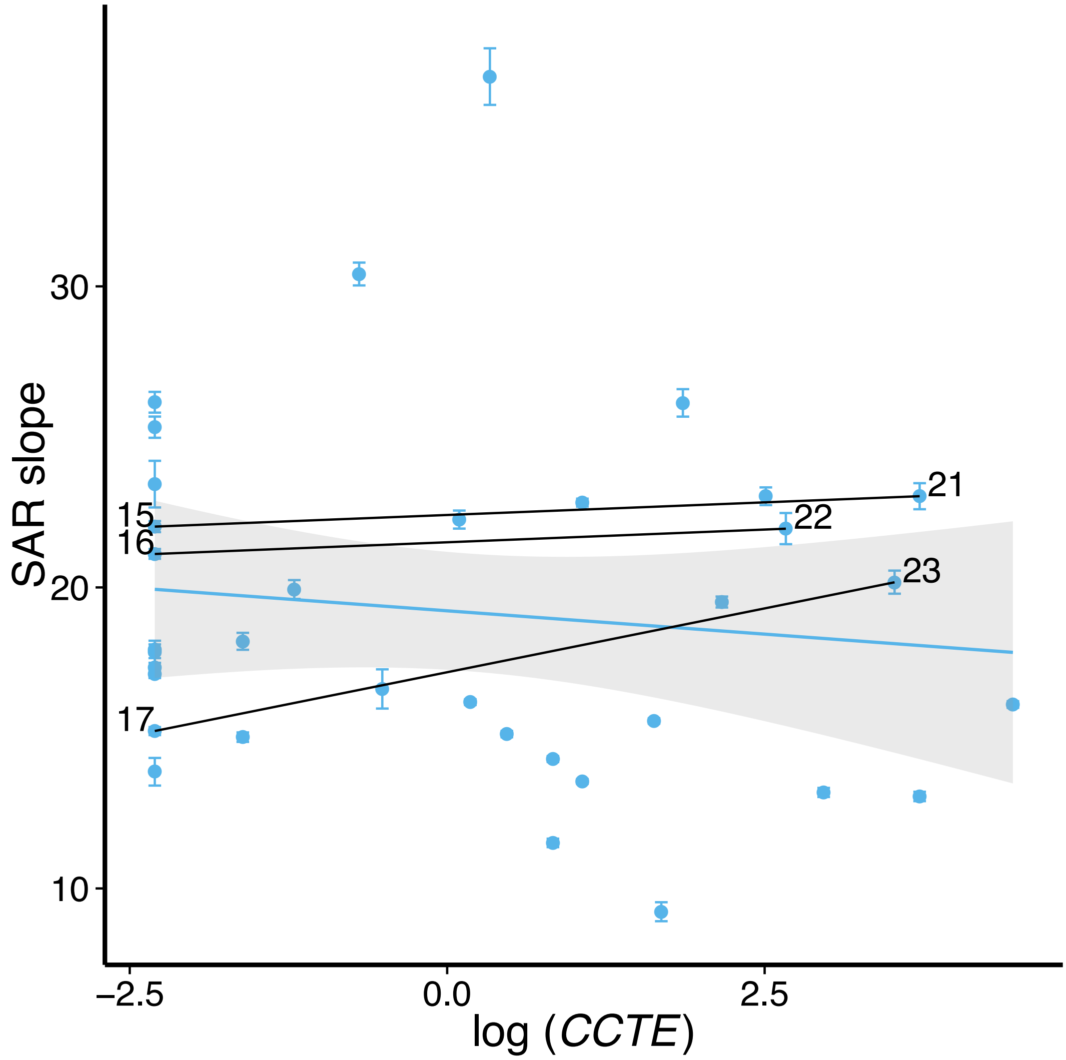

Figure 10-4. Values of z characterizing each stratum. The blue line shows a decreasing trend in z at increasing Cumulative Commercial Trawling Effort (*CCTE*) (h/km^2) when all strata are considered. Black lines show comparison between strata 15 and 21, 16 and 22, and 17 and 23.

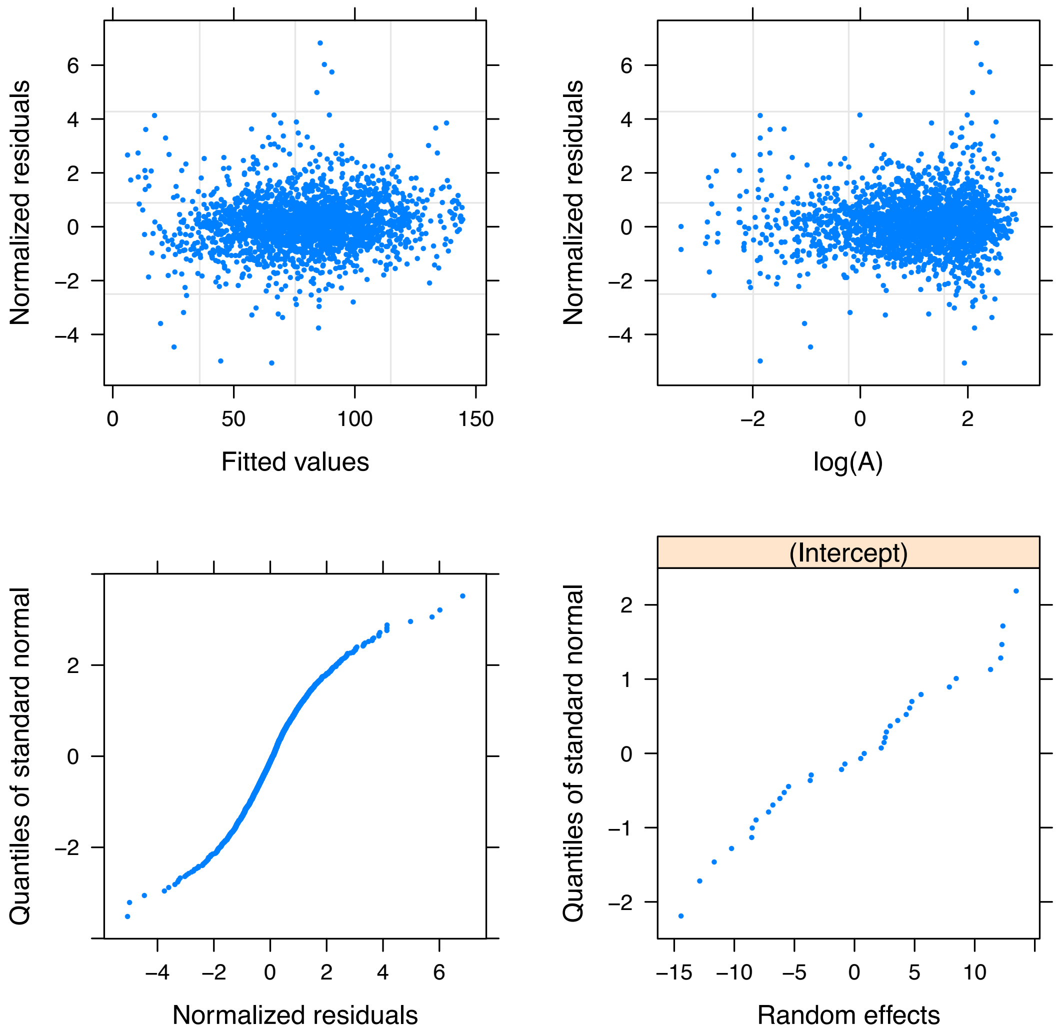

Figure 10-5. Linear Mixed Effects (LME) model's validation plots. (a) Standardized residuals versus (a) fitted values (b) the covariate *log(A),* where no strong patterns are evident. (c) Q-Q plot of the standardized residuals, suggesting residuals normality. (d) Q-Q plots for random effects (intercept), indicating that the condition of normality is met.